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CHAPTER 4

# Workgroup II Synopsis: Contaminant Fate and Effects in Freshwater Wetlands

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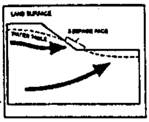
Pollution ecology is one of the few disciplines in biology that grew out of a societal need to fix a problem. The research community was forming questions as well as simultaneously developing methods, both toxicological and analytical, to address the questions in a cultural framework that demanded immediate answers.

Aquatic toxicologists wrestled with pollution issues as they developed. By establishing basic methods and sorting out different responses between ecosystem compartments, an assessment philosophy emerged that enabled us to better investigate contaminant impacts (Mount and Brungs 1967; Mount and Stephan 1967).

## CHAPTER PREVIEW





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More recently developed disciplines, such as sediment evaluation and ecological risk assessment, have benefited from the early investment of scientists who guided the development of aquatic toxicology. Some of the basic philosophical tenets were in place, and as a consequence, these more recent disciplines developed at a faster rate than those established earlier. Even in instances where the foundation was not a good "fit," it provided a starting point from which modifications could be made, increasing the chances that conceptual or methodological mistakes might be few in number or avoided altogether.

Research on wetlands did not originally focus on toxicology. Wetlands research has long been conducted by aquatic ecologists, hydrologists, waterfowl biologists, botanists, limnologists, etc., many of whom were interested in the structure, function, and biota of different types of wetlands. Management values have also figured into the equation. In the U.S., for example, federal and state agencies manage wetlands for migratory birds, endangered species, bait production, and flood control, just to list a few management values driving research. Water quality improvements resulting from implementation of Sections 402 and 404 of the Clean Water Act (CWA) have had a positive effect on wetland management. In addition, the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) also affect many aspects of wetland management. Nongovernment entities such as sportsman groups and conservation organizations also work to protect and manage wetlands.

Unfortunately, there is often a large "disconnect" between the wetlands research community and the risk assessment community regarding wetland data availability and its interpretation. There is a smaller, but nonetheless important disconnect between aquatic toxicology and risk assessment groups. When factors in risk assessments are dealt with as uncertainties, sometimes it is due to a lack of awareness that data exist outside the contaminant realm.

One of the most fundamental oversights in risk assessment is the failure to recognize that most of the remaining freshwater wetlands in the U.S. are altered from their natural state because of changes in hydrology and surrounding land use. For example, surface- and groundwater extractions and diversions for urban and agricultural water supply have affected the hydrology of many wetlands and changed their water quality, vegetation, and animal life (Thompson and Merritt 1988; Lemly 1994). Development of wetlands for other land uses has fragmented large wetland complexes into small remnant wetlands that cannot maintain their original function in water storage and supply or as habitat for biota (Frayer et al. 1989; Moore et al. 1990). Dredging and channelization for navigational purposes have disrupted the hydrologic balance necessary for riparian wetlands to effectively intercept and moderate flows and water quality degradation associated with stormwater and agricultural runoff (Lowrance et al. 1984; Philips 1989; Richardson 1994; Culotta 1995). These physical alterations constitute a chronic stress that



influences the way wetland ecosystems respond to new or added stress. On a regional and national scale, physical alterations are having a far greater impact on the integrity of wetlands than are chemical and biological threats.

In addition to recognizing and understanding that most freshwater wetlands have already been altered to some degree, it is necessary to place the risk assessment process into an ecological context. This involves identifying and integrating the principal ecosystem attributes (ecology, hydrology, geomorphology, soils) that serve to structure these wetlands and determine contaminant transport, chemical speciation, and biological exposure and effects. This chapter presents an ecosystem-based approach for evaluating impacts and risks to freshwater wetlands from chemical, biological, and physical stressors.

Major external factors such as climate, geomorphology, and soils determine the base conditions in which wetland ecosystems operate (Figure 4-1). Regional climate influences not only temperature, which mediates many biological processes within the ecosystem, but also the amount, form, and timing of precipitation. How these climatic variables are expressed in the landscape depends in significant part on regional and local geomorphic setting and soils. Of particular importance to wetlands is the way in which these external factors interact with internal wetland processes to determine the risk setting, i.e., the transport, fate, and effects of contaminants or other stressors (Figure 4-2). Wetlands provide a critical link between uplands and aquatic systems (streams, rivers, and lakes) whether the connection is across the surface or through the ground water (Figure 4-3). It is this critical linkage that in part determines the importance of wetlands as biogeochemical filters or transformers buffering flows from uplands to aquatic systems. In addition, it is this critical linkage that often places wetlands at risk and makes them an important component of many toxicological evaluations.

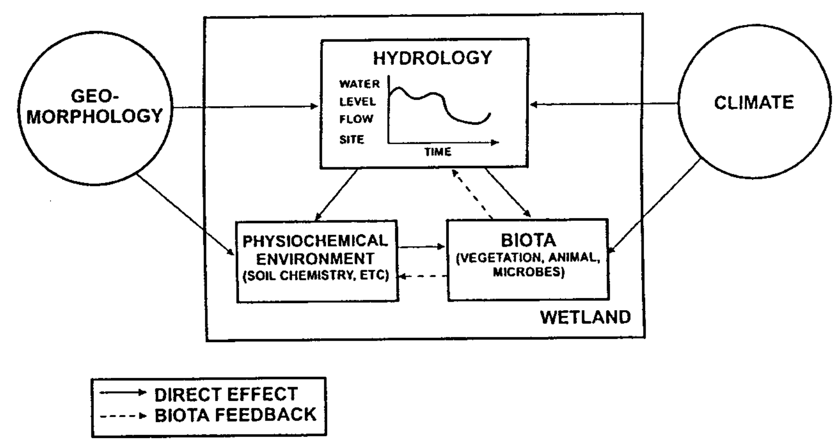


**Figure 4-1** Major external factors that determine the baseline conditions in which wetland systems operate

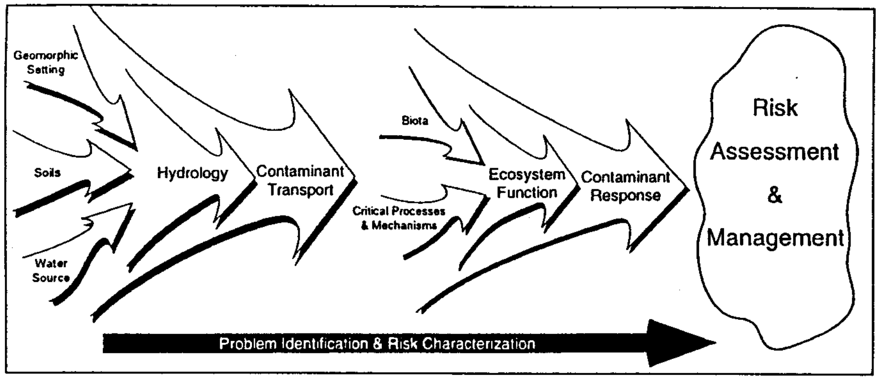


**Figure 4-2** Interaction of external factors and internal processes that determine the risk setting (potential for transport of and impacts from stressors) for wetlands

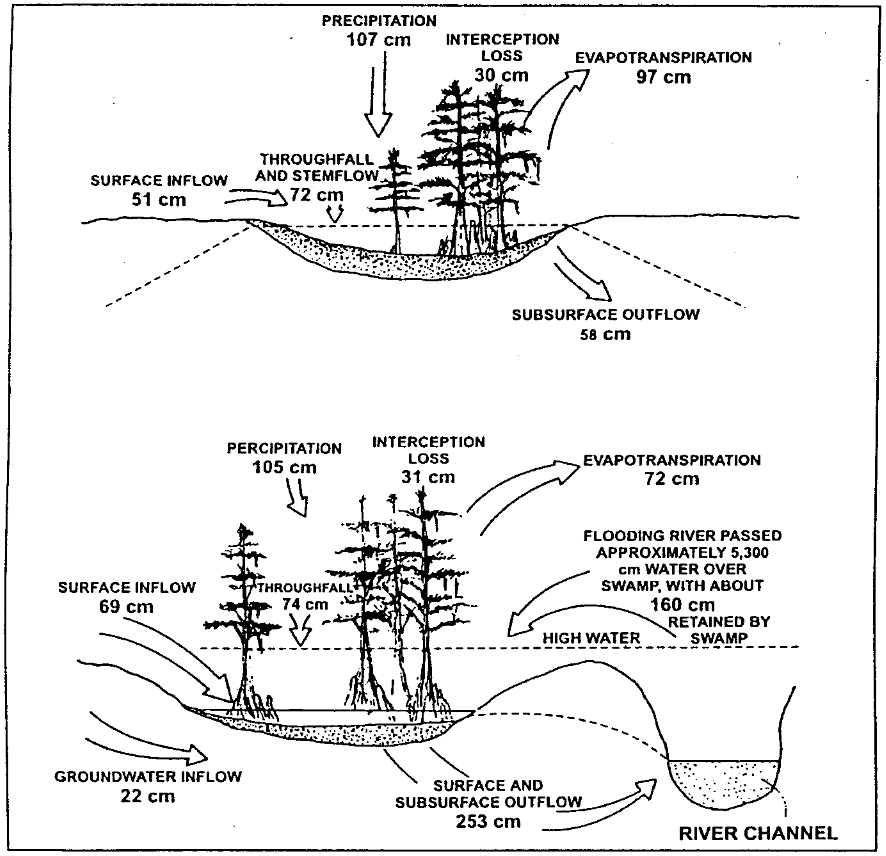


**Figure 4-3** Annual water budgets illustrating the critical hydrologic link that wetlands provide between uplands and aquatic habitats



The complex interaction of external controlling factors on wetland ecosystems, when coupled with the diverse array of internal interactions, has led to a myriad of wetland types throughout North America and the world (Table 4-1). Wetlands vary from infrequently inundated, isolated, depressional wetlands (such as some prairie potholes, Playa lakes, and vernal ponds) to large, spatially complex landscape systems (such as the Everglades, Okefenokee Swamp, and the Great Dismal Swamp). The proportion of the landscape dominated by wetlands varies from areas such as Florida and Louisiana, where wetlands are significant features in the landscape, to other areas such as the north-central section of the U.S., where wetlands represent only a small portion of the landscape.

**Table 4-1** Major types of wetlands in the United States (from NRC 1995)

| Wetland type | Distribution and hydrology | Major vegetation |
|---|---|---|
| Freshwater marsh | Widespread; seasonal to permanent flooding | Grasses, sedges |
| Tidal salt or brackish marsh | Intertidal zones; semidiurnal to fortnightly flooding | Salt-tolerant grasses and rushes |
| Prairie pothole | Northern plains states; temporary to permanent flooding; fluctuating water levels | Grasses, sedges, herbs |
| Fen | Associated with mineral-rich water; permanently saturated by flowing water | Sedges, grasses, shrubs, trees |
| Bog | Abundant in recently glaciated regions; precipitation is the principal source of water | Sphagnum moss, shrubs, trees, desmids |
| Swamp | Prolonged saturation and flooding | Cypress, gum, red maple |

## Current Practices in Risk Assessment for Freshwater Wetlands

Through various sources of regulatory guidance, wetlands risk assessment may be pursued in either a qualitative or quantitative manner (Pascoe 1993). Wetlands may be characterized hydrologically by following an engineering practice or by an analysis of physical structure (Brinson 1993). Alternatively, with respect to risk, wetlands may be characterized in an ecological or biological context, which generally yields a focus on soils and vegetation or wildlife (Federal Interagency Committee for Wetland Delineation [FICWD] 1989). For ecological risk assessments for wetlands, these independent approaches have infrequently been fully integrated into the risk assessment process as it is currently practiced.

Risk assessment for wetlands may focus on noncontaminant as well as contaminant issues (Leibowitz et al. 1992; Pascoe and DalSoglio 1994). In practice, physical, chemical, and biological stressors generally impact wetlands simultaneously. In evaluating the role of these stressors, various issues must be resolved during the problem formulation and risk characterization phases of the ecological risk assessment process (USEPA 1992, 1998). To ensure that the risk assessment meets the risk manager's goals, management and policy input must be clearly stated prior to the risk analysis activities associated with exposure and ecological effects assessment. Problem formulation includes the resolution of interrelated questions on data interpretation (performance-based versus criteria-based practices) and the distinctions among risk analysis (complete process), risk assessment (determining risk), and risk management (dealing with risk).

## Performance-based and criteria-based practices

Performance-based practices are those that specify design-focused evaluation of wetlands; for example, a naturally occurring or constructed wetland may be considered an effective remediation measure if it decreases heavy metal concentrations in mine tailings runoff by 80%. Criteria-based evaluation practices frequently assess the wetland water quality function by some numeric value developed as a consequence of a regulatory objective; for example, water discharged from a remediation wetland must meet the drinking water standards for heavy metals. Evaluations of wetlands may integrate these concepts to varying degrees (Hammer 1990) with the regulatory context that may be associated with the risk assessment. Regardless of the data sources being used in the risk assessment (e.g., historic data or data derived from designed studies), technical data collections must be applied within the data quality objectives that are developed from either performance-based or criteria-based needs.

## Functions versus values and threats versus impacts

The relationships among risk assessment and risk management activities relative to wetlands may be markedly different, especially within the context of a technical characterization of wetland "functions" versus a more risk assessment-like consideration of wetland "values." The roles these potential differences play in evaluating "threats" and "impacts" of anthropogenic activities on wetlands are subsequently dependent upon clear distinctions being given to all these terms.

Wetlands generally are considered to have functions related to hydrology, water quality, and habitat. Hydrologic functions are generally characterized by capacity and input, which may define a wetland as a water source or water sink. Water quality functions are generally focused on physical (e.g., sedimentation and stabilization) or chemical (e.g., denitrification or contaminant removal) characteristics of surface water and ground water within the wetland. Habitat functions of wetlands may be nested with subsets of functions related to biological processes such as decomposition, biological productivity, and biogeochemical processing, but these all directly



reflect the biological components of wetland structure (Adamus and Stockwell 1983; Adamus et al. 1987; Brinson 1993). For example, wetland vegetation clearly is critical and plays a major role in maintaining biodiversity and species-critical functions such as reproduction, feeding, and dispersal. Wetland values refer to the benefits obtained by society from wetland functions; for example, wetland values would include flood control and the economic benefits derived from that wetland function (Kentula et al. 1992; Richardson 1994). While the distinction between functions and values is not without technical disagreement, wetland functions are relatively easy to address within risk analysis, but wetland values are better characterized as assessment endpoints wherein societal and policy influences become critical to their definition.

For wetland risk assessments, these terms from wetland science and related wetland assessment disciplines must be clearly defined and distinguished as assessment endpoints or measurement endpoints, if the wetland scientist and risk assessor are to communicate effectively with resource managers. Similarly, the concepts of threats and impacts to wetlands must be established within an ecological risk setting. Within a risk assessment context, threats are considered sources of undesirable disturbance or activities associated with potential adverse effects (Kentula et al. 1992), while impacts are anthropogenic activities (planned or unplanned) or sources that are associated with effects that resource managers may characterize as "adverse." Risk management objectives must be adequately characterized in order to clearly identify measurement endpoints that will distinguish (or eliminate from further analysis) differences between wetlands at risk and their reference environments. In order to develop cost-effective risk analysis programs, the concepts of function and value as well as threat and impact must be consistently defined by wetland scientists and those in the risk assessment community. Wetland risk assessors and managers must clearly define assessment endpoints to ensure that their risk assessment needs are supported by the measurement endpoints that drive the technical activities of ecosystem sampling and measurement (USEPA 1992).

## Procedures for assessing and evaluating wetlands

Technical activities that support wetlands evaluation have been developed by state and federal governments (Table 4-2). These technical activities include guidance designed with wetlands as a chief focus or consider components within wetlands that make the guidance equally amenable to the wetland evaluation process (e.g., biological assessment methods for evaluation of surface water). The U.S. Army Corps of Engineers (USACOE) and U.S. Environmental Protection Agency (USEPA) have both developed guidance for evaluating wetlands. Similarly, the Natural Resource Conservation Service (formerly Soil Conservation Service [SCS]) has developed procedures for identifying wetlands for compliance with the "Swampbuster" provision of federal wetland conservation legislation (FICWD 1989). Similar technical approaches developed by states are also available and may be applicable when the assessment activities fall under the jurisdiction of state regulatory offices

**Table 4-2** A relative comparison of the applicability of technical approaches to risk or risk-related wetland assessment

| Guidance | Problem formulation | Risk analysis: Exposure assessment | Risk analysis: Effects assessment | Risk characterization: Integration | Risk characterization: Uncertainty analysis | Risk characterization: Risk summary | Risk characterization: Ecological significance |
|---|---|---|---|---|---|---|---|
| Wetland delineation | + | – | + | + | – | – | +/– |
| Hydrogeomorphic classification | + | – | +/– | + | – | – | +/– |
| Wetland evaluation technique | + | – | + | + | – | – | +/– |
| Avian richness evaluation method | + | – | + | + | – | – | +/– |
| Synoptic wetland assessment | + | – | +/– | + | – | +/– | +/– |
| CERCLA risk assessment | + | + | + | + | + | + | + |
| Natural resource damage assessment | + | + | + | + | + | + | + |

+: step explicitly included in process
–: step not explicitly included in process
+/–: step can be included depending upon case-specific implementation

(Adamus 1993a, 1993b). While the methods and guidance summarized in the following sections are not exhaustive, they are representative of the technical methods that are currently available.

### *Wetland delineation*

The Federal Manual for Identifying and Delineating Jurisdictional Wetlands (FICWD 1989) was the first effort to bring together the 4 federal agencies that had primary responsibility for oversight of wetland management or enforcement of wetland regulations (USEPA, U.S. Fish and Wildlife Service [USFWS], USACOE, and SCS). Support for this manual was withdrawn in 1991 by Congress and since that time, USEPA, USACOE, and USFWS have agreed to accept the 1987 manual developed by the USACOE for delineating wetlands. The Natural Resources Conservation Service has developed its own manual to deal with lands that fall under the Farm Bill, a specific federally legislated funding and assistance authorization. The 1982 USACOE manual provides technical guidance to establish physical boundaries of wetlands and uses the following definition (USACOE 1982):



> Those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas.

Assessment of the status of wetlands has been assigned to the USFWS by Congress. They inventory wetlands periodically, make maps of wetland areas nationwide, and make reports directly to Congress. However, their definition of wetlands does not fit precisely that of the jurisdictional definition (Hook 1993).

Currently, wetland delineations are a critical part of the wetland assessment and monitoring process. Depending upon the intensity and the data quality objectives associated with any designed wetland evaluation, various levels of effort determine the extent to which a wetland is characterized. Jurisdictional wetland delineations completed following guidance from the interagency manuals currently available (USACOE 1987) include evaluations of soils (hydric soils), vegetation, and hydrology as part of the regulatory process under the Comprehensive Wetlands Act. These methods of evaluation may be pursued with various levels of efforts, which have been categorized as screening, intermediate, and advanced applications. For soils, screening-level evaluations may be pursued using existing data; alternatively, a field survey may be conducted to collect site-specific data focused on the soils occurring in and around a wetland at risk. Similarly, vegetation may be evaluated within the bounds set in the study's design, which may include a cursory review of existing vegetation data for the wetland or an exhaustive field survey wherein a thorough identification and mapping of all plant species is accomplished as part of the delineation effort. In completing the hydrology evaluation within a wetland delineation, it should be noted that hydrologic characteristics are the least exact and most difficult to establish in the field, primarily because of the temporal variations (daily, annual, and seasonal) in water levels. In general, the 1987 delineation manual considers wetland hydrology present when soils are saturated to the surface or inundated sometime during the growing season for 7 or more consecutive days. The geomorphic setting influences the interpretation of soil saturation; hence, the drainage class of the soils must be clearly identified.

### *Hydrogeomorphic classification*

In addition to the wetland delineation process, USACOE and wetland scientists supporting their Waterways Experiment Station (Vicksburg, MS) are developing a procedure for assessing the functions of wetlands that may be useful in the risk assessment process for wetlands (Brinson 1993).

Wetland ecosystems in the U.S. occur under a wide range of climatic, geologic, geomorphic, and hydrologic conditions. This diversity of conditions makes the task of assessing wetland functions difficult because not all wetlands perform functions in the same manner or to the same extent across the U.S. To simplify the assessment process, USACOE has found it useful to classify wetlands into groups that function similarly. Classification narrows the focus on the functions of a particular type of wetlands and the characteristics of the ecosystem and landscape that influence these functions.

The classification procedure summarized below is intended primarily for evaluating the ability of wetlands to perform specific functions. The benefits of classification are a faster and more accurate assessment procedure, which supports the USACOE regulatory program mandated by Section 404 of the CWA. With this regulatory application in mind, hydrogeomorphic (HGM) classification can be used:

1) to compare project alternatives,
2) to compare pre- and post-project conditions for determining impacts or mitigation success,
3) to provide guidance for avoiding and minimizing project impacts, and
4) to determine mitigation requirements.

Hydrogeomorphic classification is modular in its design, and when compared to the risk assessment framework, its hierarchical format should make it easily adaptable to a variety of wetland risk assessment needs, including planning and management of various regulatory situations that involve the assessment of wetland function.

Wetland functions are the actions that are naturally performed by wetlands which result from the interactions among the structural components of a wetland—such as soil, detritus, plants, and animals—and the physical, chemical, and biological processes that occur in wetlands. A process is a sequence of steps leading to a specific end; for example, from a biological perspective, the microbially mediated process of denitrification occurs in many wetlands and leads to a relatively simple wetland function of nitrogen removal. Complex functions resulting from the interaction of structural components and multiple physical processes can also be identified; for example, the physical processes of overbank flooding, reduction of water velocity, and the settling of suspended particulates interact with physical structures and result in the wetland function of particulate retention.

Hydrogeomorphic classification categorizes or groups wetlands on the basis of 3 fundamental characteristics: geomorphic setting, water source, and hydrodynamics. At the highest level of the classification, wetlands fall into 1 of 5 basic HGM classes: depression, slope-flat, riverine, fringe, and extensive peatland.

Hydrogeomorphic classification's hierarchical design can be applied at a regional level to narrow the focus of the classification. For example, ecoregions identified by Omernik (1987), Bailey (1994), or Bailey et al. (1994) may be used as the next filter in the classification scheme. These ecoregions are defined in part by climatic, geologic, physiographic, and other criteria and provide a convenient starting point for applying the classification at a regional level. Within a region, any number of regional HGM subclasses can be based on landscape factors such as geomorphic setting, water source, soil type, and vegetation. While the number of regional



subclasses depends in part on the objectives established early in the assessment process, within an ecoregion the number of wetland subclasses will reflect the diversity of condition in a region. Regional subclasses provide a scale that optimizes the assessment process within the context of the USACOE regulatory program for Section 404 of the CWA.

The assessment procedure applies the concepts of HGM classification, functional capacity, reference domain, and reference wetlands. These concepts fit well within the risk assessment framework outlined by USEPA (1992, 1998) and occur within the HGM procedure as 3 phases: characterization, assessment, and application.

The characterization phase includes the

1) definition of assessment objectives;
2) characterization of the proposed project, the wetland ecosystem, and landscape context;
3) screening for "red flag" features;
4) identification of wetland assessment areas within the project area on the basis of HGM classification; and
5) physical separation and potential project impacts.

Clearly, these elements of the characterization phase of HGM classification are consistent with the elements found in the problem formulation phase of the ecological risk assessment framework. These elements occasionally have been considered in wetlands risk assessments that were completed prior to the development of HGM classification, but clearly current guidance should make that process more readily available to the risk assessment community (Brinson 1993).

The assessment phase of the HGM procedure measures the ability of a wetland to perform in terms of "functional capacity." Functional capacity is based on structural components and physical, chemical, and biological processes of the wetland. Depending upon the function, functional capacity and the extent to which it is displayed depend upon interactions between the wetland and the surrounding environment. For example, consider the floodwater storage function performed by some wetlands and the concept of functional capacity. A wetland's inherent capacity, or the theoretical capacity of a riverine wetland to store a volume of overbank floodwater in this example, depends on structural as well as physical characteristics that determine the wetland's storage capacity; however, the functional capacity, or actual amount of floodwater stored in the wetland, depends on the ability of the watershed to generate overbank floods, the state of soil saturation, and the timing of overbank floods. Watershed characteristics such as the size of the watershed, the intensity and duration of precipitation in the region, runoff coefficients of the watershed, and the location of control points in the stream above and below the wetland will influence the wetland's functional capacity, which could differ significantly from its inherent capacity.

Functional capacity of a wetland is described by the functional capacity index (FCI), which is a ratio of the functional capacity of a wetland under an existing or predicted condition to the functional capacity of a wetland under "attainable conditions." Attainable conditions are defined as the conditions under which the highest, sustainable level of functional capacity is attained across the suite of functions that wetlands in a reference domain naturally perform. The "reference domain" is simply the group of wetlands for which an FCI is developed. The reference domain normally is a regional HGM subclass, but depending on assessment objectives, it could be composed of a larger or smaller number of subclasses and geographic extent. For example, if the assessment objective is to compare a subclass of wetlands in the watershed, the reference domain would include all wetlands in the subclass in the watershed. Attainable condition, or the highest sustainable level of functional capacity, would ideally occur in wetlands that occur within landscapes that have not been subject to anthropogenic disturbance associated with long-term effects. When undisturbed wetlands and landscapes do not exist or cannot be reconstructed from historical data, attainable condition is assumed to exist in the wetland ecosystems and environments that have been subject to the least amount of anthropogenic disturbance.

Functional capacity indices are based on an assessment model that defines the relationship between the ecosystem- and landscape-scale variables and functional capacity. The condition of a variable is measured directly or indirectly using indicators that correspond to specific variable conditions. Variables are assigned an index ranging from 0.0 to 1.0, based on the relationship between variable condition and functional capacity in the reference domain that is established using reference wetlands. A "reference wetland" is a group of wetlands that represent the range of conditions that exist in wetland ecosystems and their landscapes in the reference domain. The range of conditions include those resulting from natural processes (succession, channel migration, erosion, and sedimentation) and anthropogenic disturbance.

Reference wetlands and their environments serve as the basis for scaling and calibrating variables in assessment models. The relationship between variable condition and functional capacity in the reference domain is established using empirical data, expert opinion, best professional judgment, or a combination of these options. The relationship is formalized by using logical rules or equations to derive an FCI ranging from 0.0 to 1.0. An FCI of 1.0 corresponds to the level of functional capacity that exists under attainable conditions for the reference domain, while an FCI of 0.0 reflects the absence of functional capacity. Functional capacity indices then provide measures of a wetland's capability to perform a function, relative to similar wetlands in the region.

As a result of the wetland assessment process, FCIs can subsequently be applied in various ways during the application phase. Functional capacity units (FCUs) can be calculated by multiplying an FCI by the area of wetland it represents. Once the functional capacity of a wetland area is expressed in terms of FCUs, a number of the comparisons critical to regulatory permit review processes can be made; for example, comparing the same wetland area at different points in time (e.g., pre/post project conditions), comparing wetlands in the same HGM class at the same point in time, and comparing wetlands in different HGM classes at the same point in time.



### *Wetland evaluation technique*

Various monitoring programs have focused on wetlands, and these monitoring and evaluation programs have developed technical methods that are amenable for use in assessing risk. Monitoring activities are used to account for temporal influences that may change risk through time (e.g., seasonal influences on exposure). While various qualitative and quantitative approaches to wetlands monitoring have been developed, these can be grouped into categories based largely on their geographic focus, i.e., the extent of spatial coverage ranges from the individual site to the watershed. Methods designed for application to individual sites, like those considered in the wetland evaluation technique (WET) and its various modes of implementation (Adamus et al. 1987), are focused on qualitative and quantitative approaches to wetland assessment for relatively small spatial areas.

The WET assesses wetland function in terms of social significance, effectiveness, and opportunity and uses predictors of wetland function, i.e., physical, chemical, or biological processes. These are similar, if not identical, elements common to risk assessment as it is presented in current guidance. The WET and similar approaches are generally qualitative but may reflect a limited amount of field investigation as part of their contribution to wetland risk assessment. Wetland functions routinely evaluated in WET include

- groundwater recharge,
- nutrient removal,
- sediment retention,
- groundwater discharge,
- nutrient transformation,
- toxicant retention,
- floodflow alteration,
- production export,
- aquatic biodiversity,
- sediment stabilization,
- wildlife biodiversity, and
- recreation and heritage.

From an ecological perspective, WET and similar methods do not measure community structure directly but assume community structure or wetland function on the basis of habitat structure (Adamus et al. 1987). While WET is generally focused on individual wetlands, it considers larger landscape associations, including watershed, topographic, and vegetation features, to develop qualitative estimates of wetland function and condition. These estimates take the form of ratings of high, moderate, or low for each function (except recreation), and in conjunction with a habitat suitability rating for fisheries, wildlife, and waterfowl, yield an evaluation for the wetland at risk. Within a given ecoregion, these qualitative estimates could be compiled to develop thresholds that could discriminate between each of the general categories of risk. While these methods are intended for individual wetlands (of limited spatial coverage), WET or similar methods have been applied to extensive wetlands characterized by many wetland types in a complex landscape. Many state regulatory agencies have applied wetland evaluation methods within their particular ecoregional setting, and as such, these methods may be available for use in wetland risk assessment (Roth et al. 1993).

### *Habitat evaluation procedures and their applications to wetlands*

Evaluation of wetland habitats for wildlife relies on methods developed by the USFWS as habitat evaluation procedures (HEPs) (USDOI 1980). Habitat evaluation procedures use individual species models identified by habitat suitability index (HSI) models to generate a composite of key species within a habitat, but only a limited number of HSI models are available for application to wetland risk assessment. While past criticism has focused on HEP's species-level orientation as opposed to a community-level orientation, its application to wetlands risk assessment should be considered, especially if regulatory drivers fall along single-species lines (e.g., threatened or endangered species in critical wetland habitats). Given criticisms of HEP and similar assessment methods, alternative technical methods are being developed, including community-level metrics focused on bird community structure.

### *Avian richness evaluation method*

The avian richness evaluation method (AREM) is one of the first rapid methods to be developed for assessing biodiversity (Adamus 1993a, 1993b). Without requiring extensive user knowledge of birds, it comprehensively addresses wetlands bird diversity and can be modified to predict diversity of other animal groups. The AREM does the following:

1) assigns a score to each evaluated wetland, which represents the number of bird species that could occur in the wetland multiplied by an estimate of the suitability of the wetland for each species;
2) creates a list of species likely to occur in the evaluated wetland that can be combined with lists predicted for other wetlands to identify minimum combinations of wetlands that will provide habitat for all bird species in an area; and
3) tallies the number of species likely to occur in the evaluated wetland and their particular characteristics, e.g., neotropical migrants, uncommon species, or game species. If they desire, users can assign scores to these characteristics and use them as weights in deriving the wetland score.



The AREM was developed because some available methods (such as WET) assign scores or ratings to wetland wildlife habitat without showing for which species a wetland has been rated high or low. Knowledge of species composition is essential if one wishes to maintain biodiversity at a regional level. For example, knowing the species composition allows one to avoid sanctioning the loss of a wetland containing many narrow-niched, regionally uncommon species in exchange for creating or enhancing a wetland with a perhaps identical number of species, but whose species are mere generalists.

The AREM is intended to be used in the same situations in which HEP is now used and can be applied in addition to, or in lieu of, HEP. A problem with HEP is that the scores it assigns are based on assessments of habitat suitability for only a few presumed indicator species. Many users have noted that the subjectivity of selecting indicator species biases the results, and scientists have widely questioned the validity of assuming that 5 to 10 species can represent the needs of the usual 50 to 100 species that are present at a site. Moreover, few of the HEP species models adequately address habitat needs either at a landscape level or during nonbreeding periods, and HEP assessments are often time-consuming.

The AREM can be used to assist and document resource decisions in the following ways:

1) Performing mitigation calculations. Agencies currently spend time "cover-typing" lands that will be altered or restored where compensatory mitigation has been deemed necessary. This consists of measuring various categories of habitat before a project and estimating any shifts in area that will occur among categories as a result of the project. Areas in each cover-type category that are believed to exist both before and after the project are multiplied by coefficients, determined through the use of HEP, that indicate the suitability of each category for selected species during both time periods. In this manner, net change in habitat suitability, at least for a few selected species, is predicted. Where wetland and riparian cover types are the habitats that are expected to change, AREM might be used in lieu of (or in addition to) HEP to calculate the habitat suitability coefficients of impacted or restored areas. If nonwetland cover types are also present, AREM could be adapted as described above.
2) Diagnosing impaired wetland quality. Where wetlands are officially considered by agencies to be "waters of a state," or where they exist within certain public trust lands (e.g., national wildlife refuges), a legal need sometimes exists to determine the degree to which wetland quality has been impaired. Avian richness evaluation models alone cannot determine this, but they can assist. For example, they are useful for diagnosing the presence of contamination problems by defining which species of birds should be present in a

wetland having a particular habitat structure. If properly designed surveys then fail to find the predicted species, it raises a possibility that nonphysical (e.g., chemical) factors unmeasured by AREM are discouraging wetland use.

3) Selecting appropriate indicator species. By defining which species to expect in particular types of wetlands, AREM can assist resource personnel in selecting indicator species that are the most appropriate for monitoring water quality or physical habitat suitability. Selecting appropriate indicator species is crucial to the proper use of HEP as well as to the development of biocriteria for wetland protection and the accurate monitoring of wetland contamination.
4) Targeting habitat enhancements. Active management of wetlands will usually be most effective when it focuses on improving conditions for species with low species habitat scores, while maintaining conditions suitable for species with high species habitat scores. In combination with other considerations, AREM can be used in this manner to suggest habitat features whose enhancement will support the largest variety of species overall or of species having a particular attribute.
5) Establishing wildlife-based classification of wetland habitats. Wetland types are commonly defined by their vegetative communities. Wildlife communities or individual species also can be useful primary or secondary features in classifying wetlands for scientific or administrative purposes. Avian richness evaluation models can assist such classifications by predicting bird species associated not only with vegetation but also with other environmental factors. Statistically defined, wildlife-based classes of wetlands could be identified by applying AREM to a probabilistic sample of wetlands in a region.
6) Optimizing biodiversity protection. Agencies and conservation groups sometimes have opportunities to purchase or trade properties to enhance regional biodiversity. When biological survey data from the subject properties are lacking, AREM can be applied (during any season) to the properties to predict their avian richness, which is often the largest terrestrial component of a region's vertebrate biodiversity. Richness estimates then can be calculated from the lists of predicted species pooled from multiple wetlands to determine which combination of wetlands is likely to support the greatest species richness. This estimate can be focused further by applying constraints related to land ownership, species characteristics, management costs, or other factors. As such, AREM can provide a complimentary, local refinement of the gap analysis approach currently used for ecosystem management and biodiversity planning at state and regional levels by the National Biological Service.

To date, AREM has been applied to only one ecoregion (the Colorado Plateau), but it was designed for easy adaptation elsewhere. Depending on the situation, the up-



front investment required to adapt AREM to another region is probably on the order of 0.1 to 0.5 full time equivalent (FTE), and perhaps double that if field validation is also desired. Adaptation requires the focused involvement of a professional field ornithologist or expert bird naturalist who is experienced in nonstatistical approaches to building habitat models. The adaptation process first requires a comprehensive review of appropriate local literature, followed by construction and encoding of preliminary models, modification of the field questionnaire, interviews with local avian experts at several habitat sites, and final revision of models and the questionnaire. The optional validation process requires selecting survey sites, conducting faunal surveys, data entry, and data analysis. Once AREM has been adapted for an ecoregion and/or habitat type, evaluations of most sites can be completed in less than 30 minutes and are usually not season-dependent. No computer programming expertise is required to adapt the existing AREM software.

### *Synoptic approach to wetland risk assessment*

As indicated in the previous section, wetland risk assessment may occur at various geographic scales, ranging from individual sites to larger watershed coverages where multiple individual sites may be embedded in the larger landscape, or alternatively, a single complex wetland may exist over a large spatial area (e.g., the Florida Everglades and the bayous of Louisiana). While approaches to each scale should be consistent, the level of effort required for each extreme—individual site versus watershed level—precludes identical methods being successfully employed to characterize these wetland features.

A synoptic approach to wetland risk assessment is generally focused on larger spatial coverages, e.g., ecoregions or states, and focuses on cumulative impacts as opposed to single occurrence events. From the spatial perspective, the synoptic approach differs from WET in its routine application to wetland evaluation. Nonetheless, the synoptic approach may be applicable within a risk assessment context, if one were considering a highly heterogeneous landscape characterized by numerous embedded wetland types or developing a landscape-level risk assessment. Relative to WET, the synoptic approach to wetland risk assessment requires more cursory data input, which reflects in part the greater spatial coverage of the assessment method.

The synoptic approach is designed for use by states and at ecoregion levels and is intended to relate cumulative impacts to wetlands among areas in these larger geographic scales. It is not designed to make these relative comparisons within small spatial scales where WET may be more applicable (Adamus and Stockwell 1983; Adamus et al. 1987). The synoptic approach consists of 5 steps or phases (Leibowitz et al. 1992) (Table 4-3), but from a technical perspective, the definition of synoptic indices and the selection of landscape indicators are critical to the wetland risk assessment completed using the synoptic approach.

Overall, synoptic indices are those actual functions and values within the landscape of interest, while the landscape indicators are the actual data used to represent those

**Table 4-3** Steps in conducting a synoptic approach to wetland risk assessment

| Steps | Inputs |
|---|---|
| Define goals and criteria of the assessment | Define assessment objectives<br>Define intended use<br>Assess accuracy needs<br>Identify assessment constraints |
| Define synoptic indices | Identify wetland types<br>Describe natural setting<br>Define landscape boundary<br>Define wetland functions<br>Define wetland values<br>Identify significant impacts<br>Select landscape subunits<br>Define combination rules |
| Select landscape indicators | Survey data and existing methods<br>Assess data adequacy<br>Evaluate costs of better data<br>Compare and select indicators<br>Describe indicator assumptions<br>Finalize subunit selection<br>Conduct pre-analysis review |
| Conduct assessment | Plan quality assurance and quality control<br>Perform map measurements<br>Analyze data<br>Produce maps<br>Assess accuracy<br>Conduct post-analysis review |
| Prepare report of synoptic assessment | Prepare user' s guide<br>Prepare assessment documentation |

indices. In general, 4 generic indices are the focus of the synoptic approach—wetland function, wetland value, functional loss, and replacement potential—but each application of the synoptic approach will require that a specific set of functions be identified. Defining wetland functions and values in each synoptic assessment will require an understanding of the interactions among wetlands and the regional landscapes. In practice, each of these elements of the synoptic approach is dependent upon the particular goals and constraints acknowledged in the initial step of the process in which risk assessor and risk manager define goals and criteria of the synoptic assessment. Each step of the synoptic assessment process requires multidisciplinary inputs, which will include technical information such as identification of specific wetland types found in the area of concern and descriptions of natural settings, as well as definitions of wetland values which may be more policy-related than technical.



## Methods applicable to National Pollution Discharge Elimination System permit process

For surface waters (including inland fresh water or near-coastal estuarine and marine waters), an integrated strategy consisting of both biological and chemical data requirements has been applied by regulators and the regulated community to protect water quality beyond the technology-based requirements in the CWA. One method for measuring the biological effects of toxic effluents released to wetlands is whole effluent testing. The USEPA and the states have used the data derived from effluent testing to assess compliance with water quality standards and to establish the National Pollution Discharge Elimination System (NPDES), which sets permit effluent limitations necessary to attain and maintain those standards. Technical guidance documents designed to support the NPDES process are available and should be consulted as part of the wetland risk assessment process. Specific guidance for the application of these tools to wetlands focuses on the integration of chemical and biological approaches for evaluating water quality; chemical, physical, and biological testing requirements; use of data; setting of effluent limitations; and monitoring. For wetlands in particular, the Technical Support Document (TSD) for Water Quality-based Toxics Control (USEPA 1991b) would be applicable. The revised TSD provides an explanation of the technical support for whole effluent testing and gives detailed guidance on development of water quality-based permit limitations for toxic pollutants.

In its application to wetlands risk assessment, the effluent testing approach to toxics control for the protection of aquatic life involves the use of acute and chronic tests to measure the toxicity of wastewaters. Whole effluent toxicity tests typically use standardized, surrogate freshwater or marine plants, vertebrates, or invertebrates to measure the aggregate toxic effect of an effluent. An acute whole effluent test is typically a test of 96-h or less in duration, in which lethality is the measured endpoint. A chronic whole effluent test is typically a longer-term test, in which sublethal effects such as fertilization, growth, and reproduction can be measured in addition to lethality. Again, numerous technical guidance documents have been published that focus on these methods, and their potential application to wetland risk assessment, especially at the organismic level of biological organization, has been demonstrated (Warren-Hicks et al. 1989; USEPA 1989; Linder, Bollman et al. 1991; Linder et al. 1994).

Given the policy implications of the NPDES process, various biological assessment methods and applications developed as surfacewater monitoring tools are available "as is" or in modified form for a wetland evaluation, e.g., index of biological integrity (IBI), index of community integrity, and rapid bioassessment protocols (Warren-Hicks et al. 1989). This section briefly describes various methods and, within a risk assessment context, measurement endpoints that can be used in field surveys of wetlands. Approaches available for wetland assessments—be those qualitative or quantitative—consist of methods commonly used to monitor periphyton, plankton, macroinvertebrates, and fish in a variety of aquatic habitats. Measurement endpoints consist primarily of direct and derived measures of population and community structure, such as relative abundance, species richness, and indices of community organization (e.g., USEPA 1973, 1987; Plafkin et al. 1988; APHA 1992).

## Risk assessment practices associated with CERCLA and similar regulations

Risk assessment activities pursued under CERCLA, or "Superfund," have become increasingly well documented since the Superfund Amendments and Reauthorization Act was promulgated in 1986, and the CERCLA process for conducting ecological risk assessments at contaminated sites has been summarized in numerous publications. When implemented for wetlands, the ecological risk assessment approach completed under CERCLA (Figure 4-4) is clearly rooted in the USEPA framework approach (1992, 1994c, 1997).

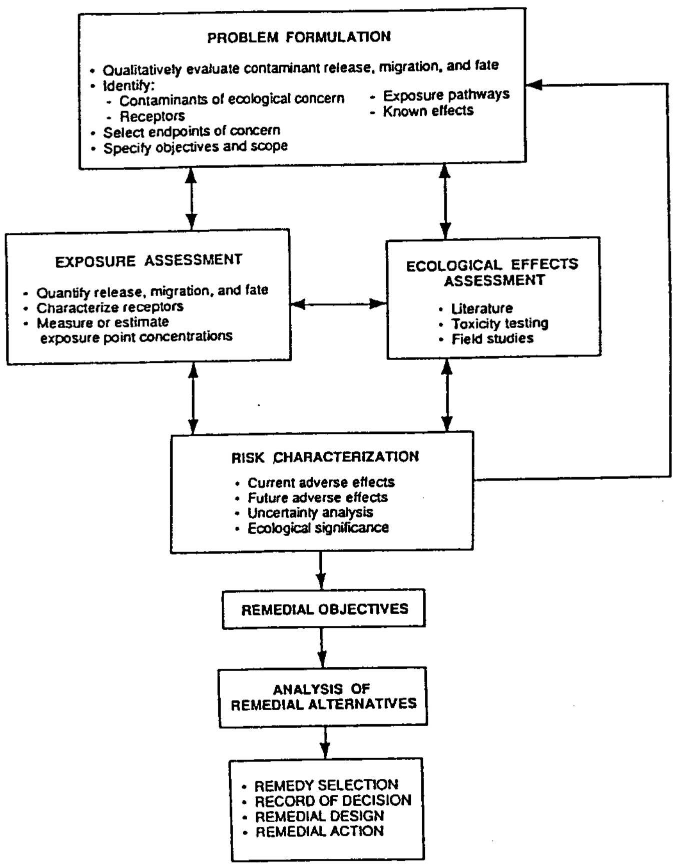


**Figure 4-4** Ecological risk assessment approach used in CERCLA or Superfund investigations



In planning ecological risk assessments for wetlands, 4 steps should be fully described to ensure that data sufficient for management needs are available for the decision-making process. As indicated in Figure 4-4, these include problem formulation, exposure assessment and ecological effects assessment (the "analysis phase"), and risk characterization.

### *Problem formulation*

Early in problem formulation, plans should be developed to address the site-specific requirements for a qualitative ecological risk evaluation. These steps in problem formulation are generally identified as

1) qualitative evaluations of contaminant releases, migration, and fate;
2) identification of contaminants and receptors of ecological concern;
3) identification of exposure pathways; and
4) selection of ecological endpoints of concern.

These steps should be carried out within an ecoregional setting; then, within a landscape setting, habitat can be used as an integrating unit for the overall process (e.g., habitat provides an ecological setting to evaluate contaminant effects associated with multiple wetlands in a particular ecoregion).

The outcome of the problem formulation phase for a qualitative ecological risk evaluation often takes the form of conceptual models, which should capture a specific set of objectives designed to address questions driving the risk assessment process (e.g., public concerns, natural resource issues) and should define the scope of the assessment required to answer these questions.

### *Ecological effects and exposure assessment*

Following the problem formulation phase and the identification of environmental resources at risk at any wetland, an ecological effects assessment should then be completed parallel to an exposure assessment in the analysis phase of the ecological risk evaluation. Within the framework process, these parallel efforts will be accomplished through a review of existing information and an extensive survey of local experts familiar with the wetland and surrounding landscape. These parallel efforts are critical first steps in assessing contaminant risks once problem formulation is complete. The exposure assessment that tracks a parallel course to the ecological effects assessment considers in detail the release, migration, and fate of contaminants of potential ecological concern (COPECs). Depending upon the existing information, the exposure assessment should more fully characterize the COPECs, as well as the ecological receptors, beyond their initial consideration in the problem formulation phase. When available within the habitat setting established in problem formulation, estimates of exposure point concentrations should also be fully characterized.

To evaluate wetland risks, an ecological effects assessment should include

1) a review and summary of historic data, as well as comparative data gathered from peer-reviewed literature and surveys of local experts;
2) a review and summary of adverse biological and ecological effects associated with chemicals and radionuclides potentially of concern; and
3) a collection of the existing field survey information for the wetland (e.g., monitoring data on wildlife or previous wetland evaluations).

### *Risk characterization*

From some perspectives, an ecological risk assessment may be considered an integrated evaluation of biological effects, derived through measurements of exposure and toxicity. From an ecotoxicological perspective, however, exposure and ecological effects assessments are complex, interrelated functions that yield estimates of risk associated with environmental contaminants in various matrices sampled at a site. Within the risk characterization phase of a qualitative evaluation of ecological risks, the outputs from the exposure and ecological effects assessments are integrated. In screening-level efforts, the integration relies heavily on strength-of-evidence arguments developed on the basis of the existing information for the facility or site. While screening-level efforts and comprehensive studies supporting the more quantitative applications of the ecological or ecotoxicological risk assessment approach differ with respect to levels of effort involved with their development (e.g., time or budget constraints), risk characterizations within any ecological risk assessment should include:

1) an evaluation of current and potential adverse biological or ecological effects,
2) an identification of the uncertainties associated with the risk characterization, and
3) an evaluation of the ecological significance associated with the contaminants or the physical disturbances associated with contaminant-related facility or site management.

In the past, risk assessments for wetlands under CERCLA were often completed as part of groundwater and soil contamination evaluations completed within the risk assessment process for a particular site; such efforts, however, may not capture the characteristics of the wetland within an ecological context. For example, groundwater evaluations completed in lacustrine, palustrine, or riverine wetlands frequently provide data sufficient for the groundwater risk assessment but may inadequately characterize the ecological context within which the ground water occurs.

As one approach to risk assessment for wetlands, guidance under CERCLA was designed to be flexible and implemented with varying degrees of effort, depending upon the landscape setting of the wetland at risk. The ecological risk assessment activities could range from being qualitative yet extensive efforts consistent with the current state of the science to comprehensive projects requiring multidisciplinary



teams of applied ecologists, research scientists, and hydrologists (see Warren-Hicks et al. 1989; USEPA 1989, 1991a, 1992, 1994a). The planning and implementation of an ecological risk assessment based upon an integrated approach for terrestrial and freshwater habitats will be discussed briefly in this section, with particular focus being given to the approach as it relates to wetlands located in various ecoregions throughout the United States.

Regardless of the regulatory and political settings, from a technical perspective, wetland habitats at risk may be evaluated using an integrated approach for ecological risk assessment that has been developed and built upon a framework originally designed for hazard assessment (Warren-Hicks et al. 1989; Suter 1991). An integrated or "ecological triad" approach evaluates risk on the basis of biological and physicochemical data in their ecological contexts (Pascoe and DalSoglio 1994; Pascoe et al. 1994; Linder et al. 1994) to evaluate risks. Whenever possible, quantitative tools are used to describe these relationships. Simply stated, the elements within the ecological triad integrate biological (including toxicological), physical, and chemical (including contaminants) information within an ecological framework (Figure 4-5). In addition to risk-driven questions (e.g., acute or chronic effects) related to chemicals or radionuclides, the analysis of risks to biological resources and ecological systems also considers indirect effects associated with contaminant exposures. Noncontaminant-related effects like physical alteration of habitat are regarded equally with contaminant effects in the integrated ecological triad approach, especially when remediation, restoration, or land-use alternatives are evaluated within the context of ecological risk.

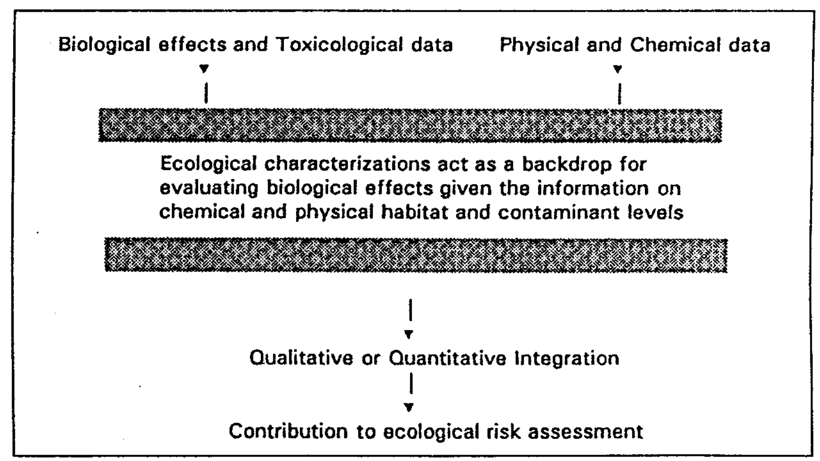


**Figure 4-5** Sources of information (biological and toxicological, physical and chemical, and ecological) that contribute to ecological risk assessment

In order to implement any ecological risk assessment, the existing regulatory guidance available from federal and state governments must be considered early in the project's organization (USEPA 1986, 1991a, 1992, 1997, 1998). For ecological risks in wetlands, the CERCLA approach is consistent with the framework document (Figure 4-6; USEPA 1992) and may be considered an integrated evaluation of ecological effects and exposure (USEPA 1991a). Within an ecological assessment, qualitative risk evaluations should consider physical, chemical, and biological interactions associated with contaminant exposures in various environmental media, e.g., soils and surface water.

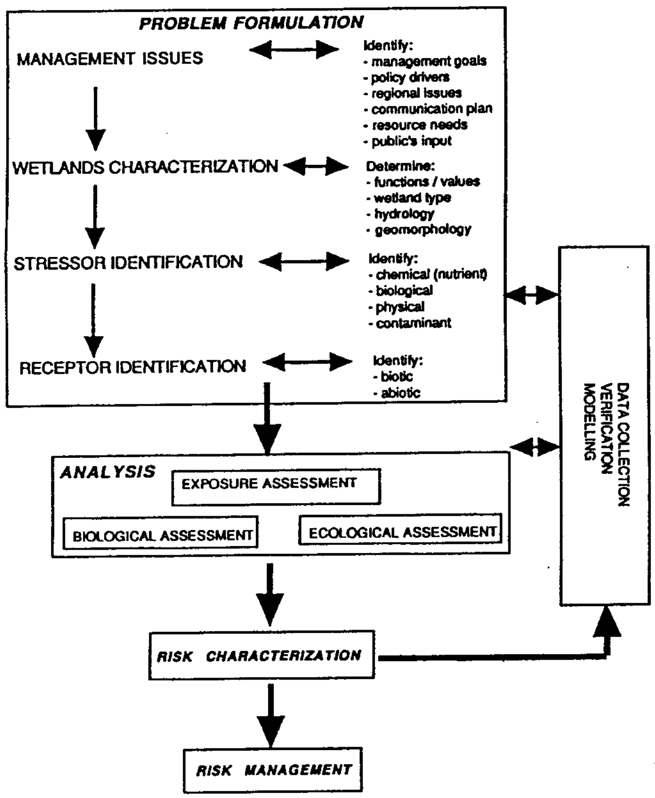


**Figure 4-6** Overview of ecological risk assessment as summarized in the USEPA Framework (USEPA 1992)

Such a qualitative evaluation of risk may be approached at various levels of effort and according to various assessment strategies. Integrated ecological risk analyses supporting wetland risk assessments are increasingly being designed under CERCLA, especially if endangered species, critical habitats, or relatively large spatial scales are of concern. Depending upon the level of effort required to satisfy the data



quality objectives for any particular wetlands, integrated ecological risk assessment activities may use

1) a "desktop" analysis of existing literature and reports,
2) a screening-level analysis, or
3) an integrated field and laboratory evaluation.

As suggested by their names, these particular risk assessment activities represent different levels of effort that are comparable to those supported through wetland guidance from the interagency manual (FICWD 1989) as well as in the developing HGM assessment process. When a fully integrated ecological risk analysis is completed for wetland risk assessments, a technically adequate data collection may be fully developed from the existing regulatory guidance and then incorporated into the risk assessment process.

Wetland risk evaluations then may range from qualitative desktop to preliminary screening efforts to comprehensive integrated field and laboratory investigations. The level of effort and implementation of the evaluation should directly reflect the data quality objectives and scope of the study that has been identified early in the problem formulation phase of the risk assessment process (USEPA 1991a, 1992, 1997, 1998). Regardless of the level of effort, chemical-based and toxicity-based approaches to risk evaluations have made significant contributions to ecological risk assessment (Parkhurst et al. 1989). From an ecological perspective, the strengths of each strategy may be combined to evaluate ecological effects and exposure within a risk assessment setting. For wetlands, exposure and ecological effects assessments are interrelated functions that will yield qualitative estimates of hazard and risk associated with environmental contaminants in various matrices sampled at a site. In the process developed for wetlands impacted by hazardous waste sites, a qualitative but extensive screening-level effort may be pursued. These initial efforts rely on existing data and include a survey of the available literature for the wetland at risk (e.g., historic wetland-specific and comparative data regarding exposure and ecological effects), as well as gathering information available through local experts familiar with historic and current status of the wetland. Ideally, if critical data gaps are apparent or more detailed information are needed to enhance risk characterizations following initial screening studies, comprehensive studies involving designed field and laboratory investigations could then address these potential sources of uncertainty (Linder, Bollman et al. 1991).

## Natural resource damage assessment and habitat equivalency analysis

Wetlands, as highly valued natural resources, may be considered in the natural resource damage assessment (NRDA) process, particularly when their functions become impaired and their values subsequently decreased. In response to the Oil Pollution Act of 1990 (OPA), the National Oceanic and Atmospheric Administration (NOAA) and other natural resource trustees (e.g., USFWS, National Park Service, and tribal governments) have developed guidance documents (e.g., USFWS 1992) on aspects of the NRDA process that may benefit the wetlands risk assessment process as it develops.

There are 3 phases of an NRDA:

1) The preassessment phase in its simplest characterization requires that the trustees determine whether an incident has occurred (under OPA, the release of oil to the environment) and whether to pursue restoration planning.
2) The restoration planning phase has its central focus on the evaluation of information on potential injuries and the application of that information to an evaluation of the need for and type of restoration.
3) The restoration implementation phase is designed to ensure that the trustees implement the developed restoration plan.

The phases of the NRDA process consider questions that are not unlike those of the risk assessor and risk manager working under CERCLA. At this time in the development history of the NRDA regulatory process, the available guidance documents suggest that the technical support for risk assessment and NRDAs will be very similar. The activities currently included in the NRDA process drive the technical support toward this similarity.

Within the restoration planning phase, the current NRDA practice addresses 2 issues:

1) a primary restoration that evaluates alternative actions proposed to return the injured resources and services to baseline or reference states, including a natural recovery option and
2) a compensatory restoration in which actions are evaluated to compensate the environment and public for the resource or services lost from the date of the incident to the recovery of the injured resources.

The type and scale of compensatory restoration is related to the type and scale of primary restoration selected, and the scaling of appropriate compensatory-restoration alternatives is primarily achieved on a service-to-service comparison of services lost as a result of the incident. When service-based cost assessment is not feasible or appropriate to the incident, compensatory restoration may also be determined through a cost analysis of lost services and gains from the compensatory restoration (see Federal Register 1995).

As in ecological risk assessment, public participation is integral to the NRDA process, particularly because that public input shapes policy in many instances. The timing and extent of public involvement in the NRDA process, and the type of documents produced at various stages of the process, fit the scope and scale of the incident in a manner distinct from yet analogous to CERCLA. In part, this stems from the past development of technical guidance by the Department of the Interior (USDOI) for assessing natural resource damages resulting from hazardous substance releases under CERCLA and the CWA. The CERCLA regulations originally applied to natural resource damages resulting from oil discharges and hazardous substance



releases. When proposed guidance is finalized under the OPA, it will supersede that currently in place for oil spill and other incidents that fall under its regulatory umbrella.

Wetland risk assessments could also benefit from the existing guidance established under the National Comprehensive Plan (NCP) for response to an incident and, similarly, could provide procedures and technical documents by which trustees could better determine appropriate restoration of injured natural resources and services (FEMA 1992). Natural resources such as wetlands are valued in terms of the services or ecological functions that they provide to other natural resources or the public, and as a result, wetlands would be potential trust resources. From an NRDA perspective, such services could be classified as

1) ecological services, or the physical, chemical, and biological functions that one natural resource provides for another (e.g., provision of food, protection from predation, nesting habitat, biodiversity); and
2) public services, or the functions that natural resources provide for the public (e.g., fishing, hunting, nature photography, education, access).

Value, as proposed for an NRDA action under OPA (Federal Register 1995), represents the amount of other goods that an individual will give up in order to obtain a good or the amount an individual will accept in order to forego the good. The total value of a natural resource or service includes direct-use values (e.g., values individuals derive from consuming or viewing a natural resource) and passive-use values (values not linked to direct use, e.g., the value individuals derive from knowing a natural resource exists). In many contexts, particularly in markets, value is represented in terms of units of currency, the commonly accepted form of exchange. However, value also can be measured in other units, including units of a resource service. In this proposed rule (Federal Register 1995), value can be measured in either units of resource services or dollar amounts. While regulatory and legal definitions may yield subtle but significant differences, values that are the focus of an NRDA and those in the HGM, WET, or synoptic approach to wetland evaluation appear similar upon initial inspection.

From a strictly technical position, the processes of wetland assessment for an NRDA and for a CERCLA are very similar. For example, potential categories of injuries include adverse changes in survival, growth, and reproduction; health, physiology, and biological condition; behavior; community composition; ecological processes and functions; physical and chemical habitat quality or structure; and services to the public, which are not unlike assessment and measurement endpoints in the risk assessment process outlined in the framework document (USEPA 1992). Although injury often is thought of in terms of adverse changes in biota, the definition of injury proposed under OPA is broader. Injuries to nonliving resources (e.g., removal of oiled sand on a beach) as well as injuries to resource services (e.g., lost use associated with a fisheries closure to prevent harvest of tainted fish, even though the fish themselves may not be injured) may be considered.

Determining exposure in an NRDA under OPA means determining whether natural resources came into contact with the oil from an incident. Early determination of exposure during the preassessment phase should focus on those natural resources or services that are most likely to be affected by an incident. In a manner similar to the analysis phase in risk assessment, an NRDA for a wetland impacted by an oil spill must determine whether the natural resource came into contact, either directly or indirectly, with the discharged oil. Exposure in an NRDA is broadly defined to include not only direct physical exposure to oil but also indirect exposure (e.g., injury to an organism as a result of a foodweb disruption).

Documenting exposure is a prerequisite to determining injury, except for response-related injuries and injuries from substantial threats of discharges. Evidence of exposure alone may not be sufficient to conclude that injury to a natural resource has occurred (e.g., the presence of petroleum hydrocarbons in oyster tissues may not, in itself, constitute an injury). Exposure can be demonstrated with either quantitative or qualitative methods. As with other elements of the NRDA process, selection of approaches for demonstrating oil exposure will depend on the type and volume of discharged oil, the natural resources at risk, and the nature of the receiving environment. For example, chemical analysis of oil in sediments, alone, may not be adequate to conclude that a benthic organism was otherwise exposed to the oil. Likewise, the presence of petroleum in fish tissue, alone, may not be adequate to link the exposure to the discharge because metabolism of the oil may blur the chemical characterization. The combination of the 2 approaches may, however, demonstrate exposure. As in the ecological triad applied in the risk assessment for the wetlands at Milltown Reservoir (see Chapter 5), exposure analysis should typically include field observations or measurements, laboratory exposure studies, transport and fate modeling, and a search of the literature. As proposed, the NRDA process emphasizes that these procedures may be used alone or in combination, depending on the specific nature of the incident. The trustees must determine the most appropriate approach to evaluating exposure on an incident-specific basis.

As in ecological risk assessment, pathway analysis is a critical component in the injury assessment phase of an NRDA. In a wetland, for example, pathways would include movement and exposure to oil through the water surface, water column, sediments (including bottom, bank, beach, floodplain sediments), ground water, soil, air, direct accumulation, and food-chain uptake. Pathway analysis includes field investigations, laboratory studies, modeling, and the reviewing literature. Again, the current practice emphasizes that these procedures may be used alone, or in combination, depending on the specific nature of the incident. The most appropriate approach to determine whether a plausible pathway exists would vary on an incident-specific basis.

To determine whether an injury resulted from a specific incident, a plausible pathway linking the incident to the injury would have to be identified, but similar to exposure, the existence of a pathway between source and target is not sufficient to conclude that injury has occurred (e.g., demonstrating that prey species are oiled can be used to document that a plausible pathway to a predator species exists, but such data do not, by themselves, demonstrate that the predator species is injured). Pathway determination can include evaluation of either

1) the sequence of events by which the discharged oil was transported from the incident and came into direct physical contact with the exposed natural resource (e.g., oil transported from an incident by ocean currents, wind, and wave action directly oils shellfish) or
2) the sequence of events by which the discharged oil was transported from the incident and caused an indirect impact on a natural resource and/or service (e.g., oil transported within a wetland by wind and wave action causes reduced populations of bait fish, which in turn results in starvation of a fish-eating bird; or, oil transported from an incident by currents, wind, and wave action causes the closure of a fishery to prevent potentially tainted fish from being marketed).

Pathway determination does not require that injured natural resources or services be directly exposed to oil. In the example provided above, fish-eating birds are injured as a result of decreases in food availability. However, trustees must always determine the existence of a plausible pathway relating the incident to the injured natural resource or service, even if the injury is not caused by direct exposure to oil.

As evidenced by the discussion of exposure and pathway analysis within an NRDA, the technical methods employed for wetland risk assessment could be identical to those supporting the NRDA process. Often, however, the language differences between NRDA and CERCLA confound an otherwise technically similar support function. Under NRDA, for example, injury quantification is the process by which trustees determine the degree and spatial or temporal extent of injuries, which supports the selection of appropriate restoration alternatives. Under CERCLA, this process does not include restoration activities. For NRDA, trustees may pursue one or more of several different conceptual approaches to injury quantification, which may be quantified in terms of

1) the degree and spatial or temporal extent of injury to a natural resource,
2) the degree and spatial or temporal extent of injury to a natural resource with subsequent translation of that change to a reduction in services provided by the natural resource, or
3) the amount of services lost as a result of the incident.

Within the context of injury quantification, the extent of injury may be expressed in terms of percent mortality; proportion of a population, species, community, or habitat affected; extent of oiling; availability of substitute services; or the spatial and temporal extent of the injury. Quantification of the total losses of wetland habitat injured by oil could be obtained by estimating the

1) total number of acres of severely oiled wetland in which vegetation is totally killed,

2) natural recovery time for severely oiled wetland,
3) total number of acres of moderately oiled wetland in which vegetation is not completely killed but the wetland has lower levels of productivity, and
4) natural recovery time for moderately oiled wetlands.

This information could then be combined to quantify the total number of "acre-years" of wetland injury to scale restoration actions.

An analysis of natural recovery, or the return of injured natural resources and services to baseline in the absence of restoration activities, may include evaluation of factors such as degree and spatial or temporal extent of injury, the sensitivity of the injured natural resource or service, reproductive potential, stability and resilience of the affected environment, natural variability, and physicochemical processes of the affected environment.

While it is beyond the scope of the present discussion to provide a detailed technical document to support either NRDA or CERCLA ecological risk assessment for wetlands, many of the technical methods applicable to the NRDA process—especially the injury and restoration assessment phases—are currently available and being used in wetland risk assessment (see section "Methods and endpoints for wetlands" and Table 4-4).

**Table 4-4** Representative technical references for aquatic and sediment biological test methods for evaluating risks in wetland habitats

| Test matrix | Target biota | Reference |
|---|---|---|
| Freshwater | Vascular plants | Wang 1991; ASTM 1997a |
| Freshwater/marine/estuarine | Algae and vascular plants | Swanson et al. 1991; ASTM 1997a |
| Freshwater | Aquatic vertebrates and invertebrates | USEPA 1990; Weber 1993; ASTM 1997b |
| Marine | Marine or estuarine invertebrates and vertebrates | Weber 1993; Klemm et al. 1994; Chapman et al. 1995; ASTM 1997b |
| Freshwater sediments | Epifauna, infauna, and vertebrates | USEPA 1994b; ASTM 1997b |
| Marine/estuarine sediments | Epifauna, infauna, and vertebrates | ASTM 1997b |

For injury assessments for wetlands, whenever practicable, procedures should be chosen that provide information of use in determining the restoration appropriate for that injury, and frequently a range of assessment approaches, from simplified to more detailed, should be considered. In general, more detailed assessment procedures may include, alone or in any combination,

1) field investigations,



2) laboratory methods,
3) model-based methods, and
4) literature-based methods.

Technical support for evaluating primary and compensatory restoration is also consistent with many of the technical procedures currently available for wetland risk assessment. Within a NRDA, trustees have the discretion to include a compensatory-restoration action as well as a primary restoration action in their restoration alternatives. Here, a scaling of compensatory-restoration actions may be appropriate. For example, in a wetland restoration action under an NRDA, a service-to-service approach may be appropriate and, in many ways, very similar to a wetlands mitigation analysis. Here, under a service-to-service approach to scaling, the appropriate quantity of replacement services is determined by obtaining equivalency between lost and replacement services after discounting appropriately for differences in habitat value. As currently proposed, trustees must use the service-to-service approach for evaluating alternatives that provide services that are of the same type and quality and are subject to comparable resource scarcity and demand conditions as those lost. This proposed "habitat equivalency analysis" has been developed by NOAA in response to OPA and is intended to be applied to the NRDA process when lost resource services are primarily of indirect human use, e.g., species habitat or biological resources like wetlands. Habitat equivalency analysis, then, may be used to scale restoration projects that replace entire habitats, e.g., wetlands, that support multiple species or that replace individual species that provide a variety of resource services. To ensure that the scale of the compensatory-restoration project does not over- or undercompensate the public for injuries incurred, the trustees must establish an equivalency between the present value of the quantity of lost services and the present value of the quantity of services provided by the compensatory-restoration project over time.

Trustees may use any reliable method for calculating interim lost value. Where a site-specific application of one of these valuation methods does not meet the reasonable cost criterion, the trustees may consider estimating interim lost value by using benefits transfer. The choice of approaches in a particular context will depend upon the types of injuries and the type of services provided by the compensatory-restoration alternative. Trustees should consider using similar methods for measuring the value of the lost services and the value of the services provided by the compensatory-restoration alternatives. If different valuation methods are used, then trustees should take steps to ensure that the variation in methods does not introduce bias.

To evaluate restoration, monitoring activities may be incorporated into the NRDA process. As in the monitoring tasks that are frequently included in CERCLA ecological risk assessments, monitoring plans within the NRDA process should address study design elements such as duration, frequency of monitoring required to evaluate progress and success, the intensity of sampling required to detect success or the need for corrective action, and monitoring of a control or reference site to determine progress and success. To evaluate success of restoration actions, performance criteria may be developed which evaluate structural, functional, temporal, and other goals. For example, an agreement to create new marsh habitat as compensation for marsh impacted by oil could be described by performance criteria including the number of acres to be created, the location, the elevation of new habitat, the species to be planted and details for planting, such as density, and the time frame in which identifiable stages of the project should be completed.

### Strengths and limitations of current risk assessment approaches for wetlands

From a technical perspective, each of the regulatory-associated practices considered above may be compared relative to the steps outlined in the USEPA framework approach (Figure 4-6). In a strict sense, no one method is best nor was any originally developed for wetlands risk assessment. Each has been molded, however, to assure their implementation for risk assessments mandated by law and regulation. In many respects, each approach summarized in this section, as well as those not included in this discussion (but available from many states and other federal agencies), requires technical support from wetland scientists, ecotoxicologists, and applied ecologists. Each approach identified in Table 4-2, for example, includes guidance for reviewing existing information for the risk assessment process or, alternatively, for designing and completing studies or surveys to address questions identified in the early phases of the risk assessment process. Similarly, each approach recognizes the importance of evaluating ecological effects, although the linkages between stressors (especially chemical stressors) and ecological effects are more thoroughly explored in some implementation plans than others. For example, explicit guidance for evaluating exposure is poorly described in some strategies for evaluating wetlands, but these guidance documents are also better developed for an analysis of physical stressors that may have impacted a wetland as a consequence of changes in land-use practice, e.g., synoptic wetland assessment versus CERCLA risk assessments. Shared limitations among all approaches include problems associated with interpreting existing information within a risk context, especially in comprehensive risk assessments that rely on statistical methods. Here, for example, data quality issues cut across all approaches, and regardless of the risk strategy employed, each shares problems related to inter-study comparisons and their interpretations, data pooling, and statistical issues related to encountered data.

Overall, the strengths and limitations of each approach considered here, as well as other approaches addressing similar risk-related questions, reflect the policy and management issues that are critical to the process, as noted in the USEPA Framework (1992, 1998). The technical support tools available for ecological risk assessment are numerous (see, e.g., "Methods and endpoints for wetlands," this chapter). But to ensure that the best available state-of-the-science is implemented to support wetlands policy and management, clear lines of communication must exist among



the policy, management, and scientific professionals involved in the risk assessment process, and the risk assessment must take in all relevant ecological information for the site.

## The Ecosystem Approach: Integrating Ecology, Hydrology, Geomorphology, and Soils of Wetlands

### Abiotic characteristics of freshwater wetlands

Freshwater wetlands represent a host of ecosystems that have an abundance of surface water some time during most years (Hook 1993). Beyond this common thread, they vary greatly in characteristics. Local climatic conditions, geomorphology, and hydrodynamics have acted over time and are still acting to create the diverse and dynamic nature of these ecosystems.

The purpose of this general overview of the relationship of abiotic factors to wetland characteristics is to point out how they can be used to quickly identify those generic functional traits that should be addressed in a wetland-specific risk assessment. However, these are general guidelines, and each site must be examined closely to determine whether expected conditions actually exist. If the information is used in this context, it should prove helpful in focusing ecological and biological risk studies on relevant issues.

#### *Climate*

For wetlands to occur, there must be excess water. It generally comes as runoff from upland drainage areas. A simple form of the water balance equation is

$$dS = P - ET - R \qquad \text{(Equation 4-1)},$$

where dS = storage, P = precipitation, ET = evapotranspiration, and R = runoff. In wetlands, ET tends to dominate this equation, and during some periods of the year, ET may exceed P so that no water is available for runoff. When $R > 0$, water runoff occurs either by overland flow and/or subsurface flow and may collect in depressions, thereby creating wetlands. Such climatic data generally are available through state and federal agencies, and they provide valuable clues as to the temporal nature of wetlands within a region (Figures 4-7a, b). Note that at Caribou, ME, R usually exceeds 0 from August through April (Figure 4-7a). Thus, one would expect the wetlands to be highly evident during dormant growth periods and less so during the growing season. In contrast, at Fort Lauderdale, FL, runoff exceeds outputs from March through November; thus, wetlands are apt to be most evident in this area during the summer months or growing season (Figure 4-7b).

#### *Geomorphology*

Geomorphology is the landscape position or geomorphic setting that accommodates the runoff and storage of water (Brinson 1993). As a consequence, geomorphology

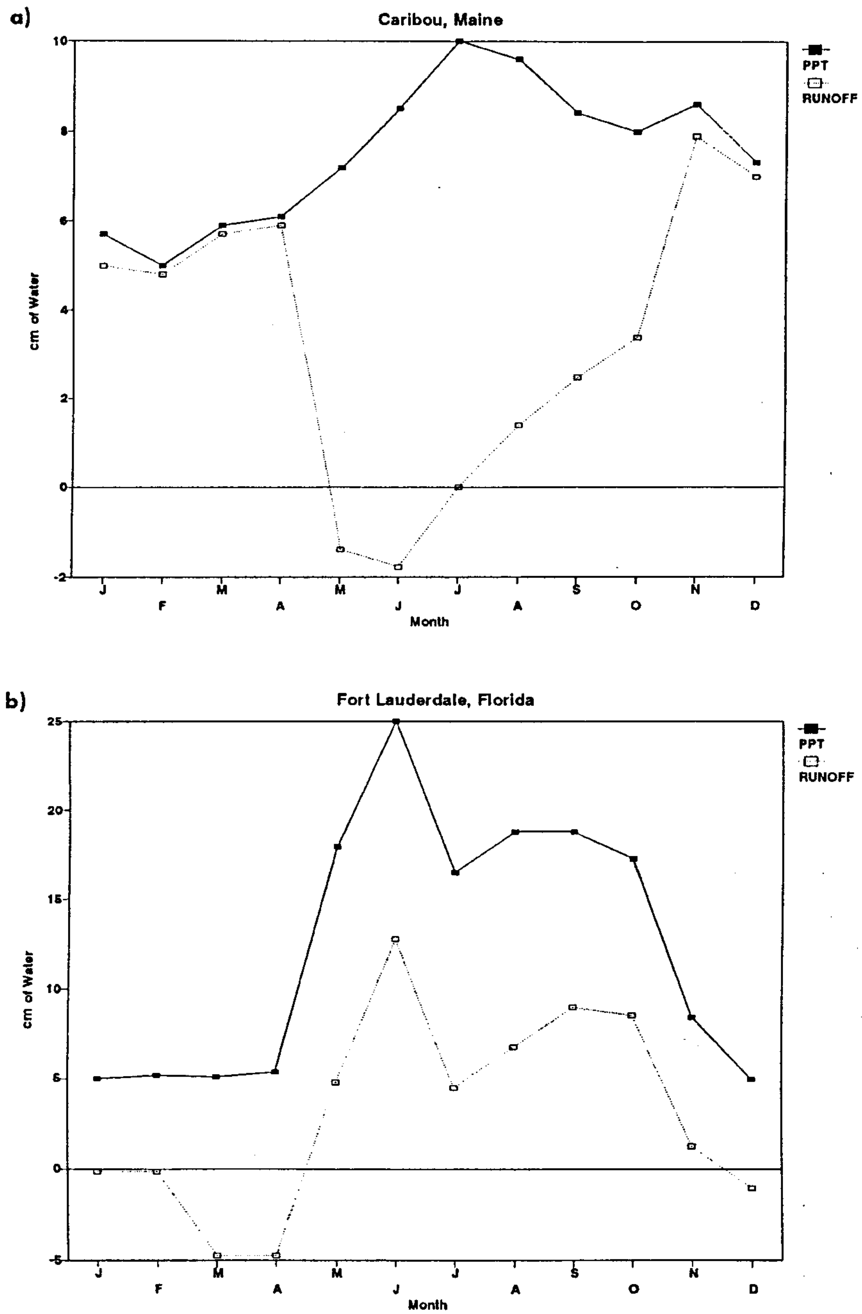


**Figure 4-7** Relationship of total rainfall in cm to runoff in cm for a) Caribou, ME and b) Fort Lauderdale, FL



is generally linked to runoff and wetland positions. There are depressional, riverine, and fringe categories of geomorphologic settings.

Depressional wetlands include such landforms as kettles, potholes, vernal pools, and Carolina bays. They frequently occur high in drainage systems; thus, they typically depend heavily upon local precipitation when compared to other geomorphic settings. In climates where ET exceeds P, depressions such as vernal pools tend to be dry much of the time, or they depend upon ground water (Zelder 1987; Brinson 1993). In climatic regions where R > 0 for a significant portion of the year, depressions may accumulate sufficient peat to develop a domed topographic relief. These types of wetlands receive their water from precipitation, not from ground water or overbank flooding.

Extensive peatlands are usually the terminal condition of peat accumulation in depressions, followed by radiating paludification. When such conditions occur, they create domed landscapes where the highest elevation receives precipitation as the sole source of water. These are generally nutrient-poor environments. They may cover large areas of land such that the peat substrate dominates the movement and storage of water, the mineral nutrition of the plants, and the patterns of the landscape (Moore and Bellamy 1974). Extensive peat formations caused by paludification across the landscape may develop surface patterns that are independent of the underlying topography. As a consequence, there is a gradient from the headwater ombrotrophic wetlands with diffuse outlets to ones further downstream with fen-like characteristics (Siegel and Glaser 1987).

Riverine wetlands form as linear strips parallel to streams but are generally separated from the stream channel by natural levees. A riverine wetland may occupy most of the floodplain in large rivers (high-order streams) in the lower coastal plain but may be very small or nonexistent in low-order streams in the Piedmont regions of the South (Theriot 1988; Hook et al. 1994). Hydroperiods range from short and flashy in low-order streams to long and steady in higher-order streams. The slope of the stream channel determines whether a given section of the floodplain is predominately erosional or depositional.

Freshwater fringe wetlands are restricted to freshwater tidal zones associated with estuaries. These types of wetlands are generally riverine (alluvial), in nature but some may be headwaters (nonalluvial). The latter occur in small drainages that feed into rivers near estuaries.

### *Hydrodynamics*

The source of water for freshwater wetlands may be precipitation, groundwater discharge, surface or near-surface inflows, or any combination of these. Many depressional wetlands receive their water from precipitation runoff. These types of wetlands occupy depressions in the landscape that are above the general water-table level (Figure 4-8a). They are generally separated from the water table by a layer of relatively impermeable soil that restricts the rate of water movement downward

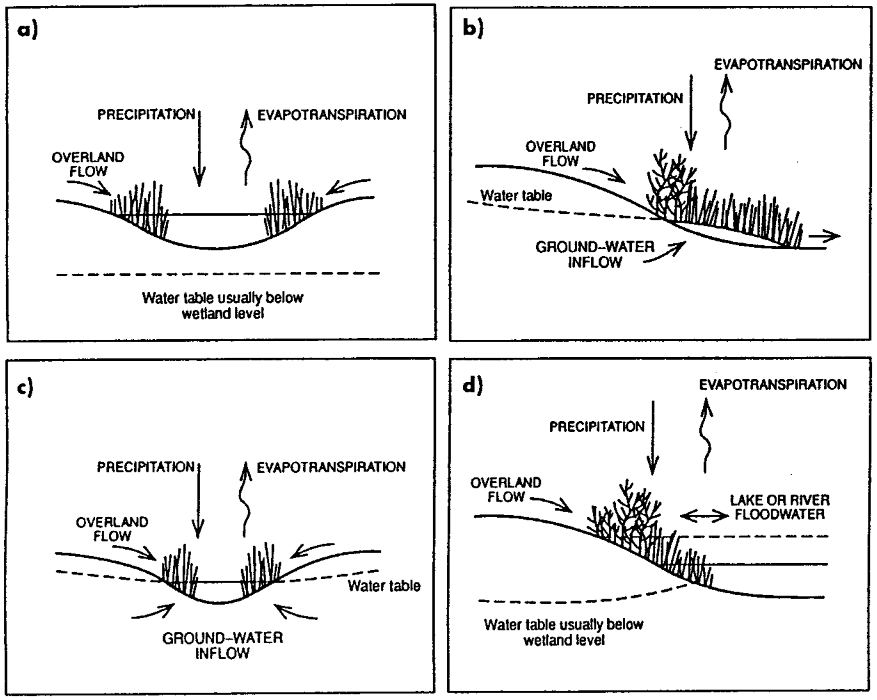


**Figure 4-8** Four major hydrologic types of wetlands in Wisconsin: a) surfacewater depression, b) groundwater depression, c) groundwater slope, and d) surfacewater slope (after Brinson 1993)

through the soil. Therefore, the dynamics of the water table are vertical. It moves up when it receives runoff and down primarily due to ET. Depressions generally have no inlets or outlets, or, if they are present, they receive or drain water only during or after storm events. They tend to be disconnected hydrologically from the surrounding landscape and the substrate below the restrictive layer. However, during high-water events, some water may spill out of the depression beyond the restrictive layer and come into contact with the substrate below. Research in Florida has shown that the cypress domes may be more interconnected than originally thought (Riekerk 1993). Depending on size, geomorphology, and regional location, they may develop distinct zonational vegetation and structural patterns in relation to the time and duration of inundation and fluctuation of the water table. Nutrient input into these systems is primarily by precipitation. On a relative scale, they tend to have low productivity. However, productivity may vary with the geology, climatic conditions, and types of soils and vegetation that develop.

Some depressional wetlands receive ground water in addition to runoff from precipitation (Figures 4-8b, 4-8c, 4-8d). If the groundwater table intersects the slope at or within the depression, water enters from below as well as from runoff. Ground water may enter wetlands or create wetlands on slopes where the water table intersects the soil surface (Figure 4-9). Such areas can best be visualized as seeps or springs. However, relatively large wetlands can occur on slopes.



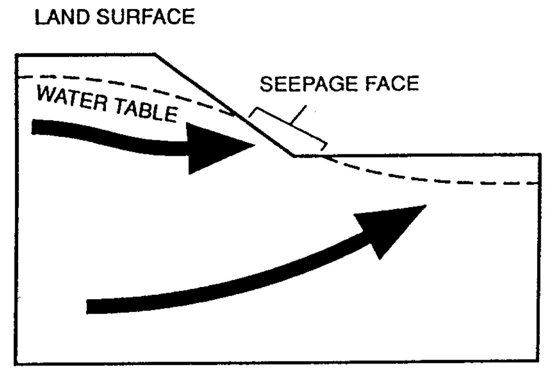


**Figure 4-9** Relationship of land surface and water table to seepage face (after Brinson 1993)

If ground water enters a wetland, it has been in contact with the mineral content of an aquifer or soil. Depending on the time of contact and the composition of the lithology, such water normally has higher mineral content than water derived from precipitation. Consequently, plant communities in wetlands that receive groundwater discharge tend to be more productive than rainwater wetlands (depressions). Furthermore, the hydrodynamics of the system are apt to be more stable than in precipitation-driven wetlands (i.e., dry-downs may not be as severe and as rapid). The dynamics of the water table in these types of wetlands tend to be vertical in relation to water inputs and outputs (Figure 4-10a).

The source of water in riverine wetlands may be from overbank flooding, ground water, and precipitation. The dominant water source is not always evident even after extensive exploration. A study in the Piedmont of South Carolina showed that a fourth-order stream received periodic overbank flooding on average about 3 times/y during the dormant season. However, during the growing season, the wetland was driven entirely by precipitation (Hook et al. 1994). In contrast, in a fifth-order stream in coastal Georgia, water came from overbank flooding during the growing season as well as the dormant season, but between major rainfall events in the watershed, precipitation and ground water influenced the wetland to varying degrees depending on topographic relief. Pesiometric studies showed that micro topography had important influences on drainage patterns and sources of water between flood events (Saul 1995).

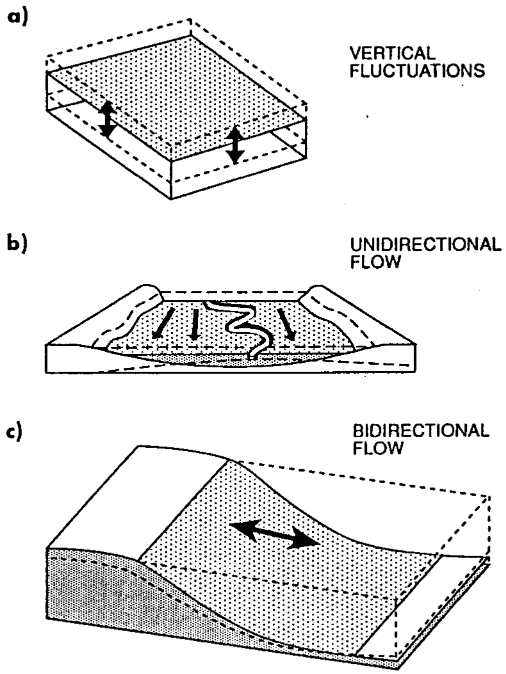


**Figure 4-10** Categories of hydrodynamics based on dominant flow pattern: a) vertical fluctuations normally are caused by evapotranspiration and precipitation, b) unidirectional flows are horizontal surface and subsurface, and c) bidirectional flows are horizontal across the surface (after Brinson 1993)

The water in a floodplain tends to flow unidirectional down stream (Figure 4-10b), but depending on topography depressions in the floodplain, it may take on vertical dynamics when the river is not in flood stage. In the lower reaches of rivers influenced by tides, the fringe wetlands may be subjected to bidirectional flows similar to those in estuaries (Figure 4-10c). The variation in hydrodynamics among wetlands and within localities of a wetland must be carefully considered if contaminant studies are to successfully identify key transport and exposure pathways to biota.

Biogenic and fluvial deposition in wetlands tend to be causally related to water flow rate (energy; Figure 4-11a). Hydrologic energy, hydrodynamics, nutrient availability, temperature, salinity, fire frequency, and herbivory are also related in a general manner to wetland type and core factors (Figure 4-11b).

When a wetland has 2 or more water sources, it can be difficult to separate their relative contributions. For riverine systems, records of time, frequency, depth, and duration of overbank flooding are necessary to evaluate the extent of individual contributions, effects of overbank flooding on the wetland, and how contaminants may be delivered, retained, and transported. Some rivers have stream gauges



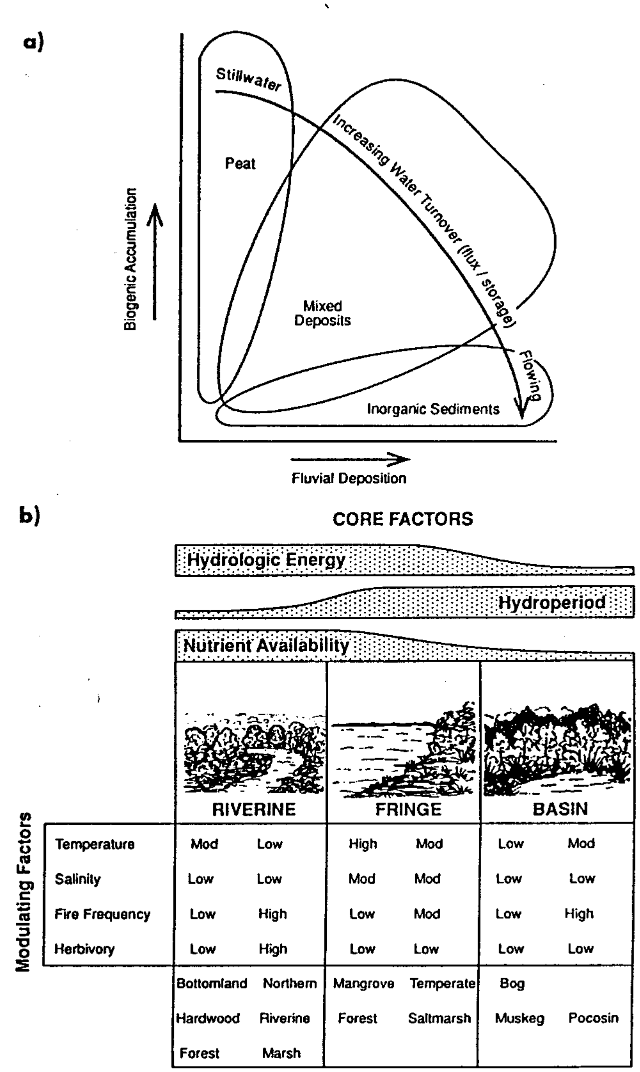


**Figure 4-11** a) Relation of water turnover to biogenic accumulation and fluvial deposition in forested wetlands b) The use of core factors and modulating factors to characterize specific types of wetlands (after Brinson 1993)

maintained by the U. S. Geological Survey (USGS) for various durations. Such records are invaluable for ecological and toxicological studies and evaluation of various wetland functions. In the absence of such records, stream flow or pesiometric (soil saturation) studies are necessary to quantify many characteristics of a wetland. Problems arise in determining how long monitoring must have occurred to be useful. For example, a 38-y record for one wetland in eastern North Carolina demonstrated that, depending on which 3- or 5-y period was selected for measurement, the site could be classified either as a wetland or nonwetland using jurisdictional criteria (W. Skaggs, personal communication).

***Use of soil surveys***

Many, if not most, counties in the United States have surveys of the soils. The surveys contain more general traits that will help determine the potential characteristics of a specific wetland. They identify soils by series and drainage class and provide information on productivity, amount of organic matter (OM), general information on the degree of soil saturation or flooding, times of hydroperiods, and occasionally the duration of hydro events. In addition, if the wetland is forested, the data bank may include information on site index for various tree species. This provides another clue to the relative productivity of the wetland (site index is the height that a tree will reach at a specified age and has proven to be a very good measure of the productivity of the site). Again, these are general traits for a soil series, but they provide the researcher with a fairly extensive array of characteristics about the wetland site in question. It is necessary to verify whether the soil information is truly indicative of the site by examining the soil profile and other salient characteristics of the site. Is the vegetation natural or has it been altered? Has the hydrology been altered by drainage and blockage of drainages? Assistance with this process can usually be found close by. The U.S. Department of Agriculture Natural Resources Conservation Service generally has offices in each county, with trained personnel who can help interpret soil survey information and sometimes assist with actual field checks. In addition, county agents and university extension personnel may be available to help interpret the data or provide guidance on where to seek help.

## Integration of abiotic factors

The salient characteristics of wetland ecosystems are embodied in the integration of local climatic conditions with geology and hydrodynamics. The results of this integration over geologic time are evident in the soils, vegetation, and biota. Thus, the wetland ecosystem is a result of the interaction of specific abiotic factors (climate, geology, and hydrodynamics) and various organisms over a long period of time. However, can abiotic traits alone be used to determine what processes and functions a specific wetland may have? The answer is—only in a general context. For instance, a depressional wetland would not be expected to be involved in carbon transport or to actively transport pollutants or nutrients out of the system. Furthermore, the system would not be expected to be highly productive, but caution is needed for the latter. If the depression has a groundwater source that is rich in nutrients, its productivity may be high; thus, it could act as an efficient buffer or transformer. Additionally, the amount of OM in the soil will influence its potential to support microorganisms for decomposition and other soil reactions.



Geomorphic settings and traits can be a practical starting point for identifying the basic type of wetland as well is its principal ecosystem functions and associated ecological significance (Table 4-5). Moreover, the relationship of abiotic factors to wetland characteristics is useful for identifying those generic functional traits that should be addressed in a wetland-specific risk assessment. The approach can be simplified to a protocol that incorporates 7 steps:

1) Determine the geomorphic setting. Is it a depression or basin, a riverine system, or a fringe wetland?
2) Determine the dominant source of water. Is it rain water, ground water, or overbank flooding?
3) Determine the dynamics of the hydrological mechanisms. Does the water table fluctuate vertically? Is it primarily unidirectional or bidirectional? Do the dynamics change with water-table level or season? Use the water balance equation and determine when R exceeds 0. This will identify the seasons or times that the water table is apt to be the highest or lowest.
4) Use all available resources, i.e., aerial photographs, maps, interviews with local people, field reconnaissance in and around the wetland, to determine if the hydrology has been significantly altered. If it has, try to determine how the alterations may have affected the hydroperiods, timing, frequency, and flow patterns that would be expected to be associated with the existing geomorphic setting.
5) Use soil surveys to determine soil series, texture, drainage class, vegetation, hydroperiods and hydrodynamics, and the relative productivity based on site index or other site productivity documentation in the survey.
6) Scout the entire area to determine the patterns of inundation, vegetation types, and vegetation densities to identify any zones or patterns that may affect how toxins may enter the wetland and how they could be influenced by open water, vegetation traits, and seasonality of hydrodynamics.
7) Determine where and how the wetland is positioned in the watershed and whether it may have been impacted by long-term chronic conditions (disturbance) of any type. Look for differences in vegetation. Does the regeneration match what is expected for the site? If not, is the regenerating vegetation more hydric or more mesophytic than is characteristic for the wetland type?

Analyzing these abiotic factors is the fist step in an ecosystem approach to wetland risk assessment. Although abiotic traits alone can provide valuable clues for targeting ecotoxicological investigations or other studies, one must also overlay information on the biology and ecology of the system in order to conclusively identify and evaluate the full range of potential issues or problems for a given assessment.

Knowledge of wetland science is necessary in order to effectively address the biotic components of wetland ecosystems in the context of risk assessment. A discussion of some of the key principles is given here to point out important factors that must

**Table 4-5** Relationships between geomorphic setting and ecosystem attributes of freshwater wetlands

| Geomorphic setting | Qualitative evidence | Quantitative evidence | Hydrologic functions | Ecological significance |
|---|---|---|---|---|
| No apparent inlet or outlet | Topographically isolated from other surface water | Drydowns frequent; water table frequently below the wetland | Retains inflow; losses mostly by infiltration or evapotranspiration (ET) | Inaccessible to aquatic life dependent on streams; endemics likely |
| Positioned on local topographic high; surface output only | Outlet may be defined by contours or intermittent streams | Drydowns frequent; water table frequently below the wetland | Temporary flood storage; outlet may overflow during high surface water or flow continuously during high ground water; outlet controls maximum depth | Wetland open to immigration and emigration of aquatic life; potential for recolonization if drydowns cause local extinctions |
| Located in marginally dry climate; variable inlets and outlets | Inlets and outlets may be defined by contours or intermittent streams | Water conductivity high = wetland is recharging underlying aquifer; if low = aquifer is supplying the wetland | Retains inflow; loss primarily by ET or infiltration; may be subject too wide for migrating fluctuations in water depth | Import and export detritus; critical habitat for migrating waterfowl; vulnerable to eutrophication and toxic accumulation due to long retention time |
| Both surface inlet and outlet; large catchment sustains marginal riverine features | Inlets and outlets may be defined by contours or intermittent streams | Water budget dominated by lateral surface flows or strong groundwater discharge | Temporary flood storage; drainage back to stream or continuously saturated | Import and export detritus; provides fish and wildlife habitat |
| Located on break in slope | Soil saturated most of the time | Chemically indicative of ground water; discharge from slope base or face | Inflow steady and continous; seasonal loss by ET; low surface storage capacity | Provides stable source of moisture; contributes to biodiversity |
| Ombrotrophic bog | Peat substrate saturated most of the time; plants indicate ombrotrophic bog; surface flows are negligible | Peat confirmed by organic content and thickness; ombrotrophy evident from low pH and ion content | Some storage of storm runoff; groundwater conservation when water table is below surface | Upland habitat scarce; species composition is unique to bog conditions |
| Rich fen | Peat substrate saturated most of the time; graminoid species indicative of groundwater supply | Peat confirmed by organic content and thickness; minerotrophy evident from circumneutral pH and high ion content | Subsurface water supply maintains saturation to surface and hydraulic gradient to maintain flow | Allows lateral movement of water without channelized flow; exhibits moderate level of primary productivity and detritus export |



**Table 4-5** *(continued)*

| Geomorphic setting | Qualitative evidence | Quantitative evidence | Hydrologic functions | Ecological significance |
|---|---|---|---|---|
| Streamside zones of intermittent streams | Headwater position; first-order stream | Flows not continous; no headwater flooding or overbank flows | Interface of landscape where ground water and surface water change to fluvial environment | Riparian zone critical to maintain buffer between the stream and uplands |
| High-gradient downcutting portions | Bedrock-controlled channel | Substrate lacks alluvium; flow may be continuous but flashy | Downslope transport is dominant feature | Scour prevents extensive wetland development |
| High-gradient aggrading portions | Substrate controlled by fluvial processes | Stratigraphy shows imbedding of coarse particles within fines | Wetland on coarse substrate maintained by upslope groundwater source | Unstable substrate in a scour-prone environment colonized by pioneer species Allochthonous organic supply |
| Middle-gradient landform | Channelized flow; evidence of oxbows and meanders consistent with fluvial processes | Flow likely continous with moderate to high base flows | Channel process establishes variation in topography, hydroperiod, and habitat interspersion on a floodplain | Interspersion of plant communities increases biodiversity |
| Low-gradient alluvial; floodplain of bottomland hardwood | As above, but in low-gradient landform | Flow continuous with cool season flooding; high suspended sediments in stream | Flood storage; conserves groundwater discharge | Major habitat for wildlife; biogeochemial activity and nutrient |
| Shoreline of large lakes | Subject to seiches; lake level controls position | Year-to-year trends in zonation follow climatic cycles; wind-generated fluctuations possible | Lake is water supply for wetland and establishes hydroperiod gradient for wetland zonation | Stabilizes shoreline; transition habitat used by aquatic and terrestrial biota |
| Coastal sea-level location | Subject to tides; sea-level controlled | Elevation relative to tides and changing sea level | Wetland is responsive to tides and sea level | Barrier to saltwater encroachment; retains sediment; nursery habitat for estuarine organisms |

be considered when identifying biological characteristics of a wetland. These characteristics may ultimately affect the direction of the risk assessment as well as the effectiveness of subsequent risk management.

## Biological processes and ecosystem functioning

In addition to the complexity introduced by the myriad of interactions of external factors, differential biotic responses to these external factors also yield a complex set of interactions among the biota (organisms, species, populations, communities), the critical processes they perform (photosynthesis, microbial action, decomposition, etc.), and the way these organisms and their processes are expressed through ecosystem functions (production, biomass accumulation, biogeochemical processes, etc.). To a large extent, the complex structure and function of wetlands reflect the divergent properties of their biota. Most wetlands are dominated by a flora of vascular plants that are adapted to a greater or lesser extent to flooded conditions, but that are, in most respects, structurally and physiologically similar to their terrestrial ancestors. Yet, wetlands may also have features similar to deepwater aquatic ecosystems, including sediment biogeochemical and biotic processes mediated through predominantly anoxic conditions and aquatic food webs of algae, invertebrates, and vertebrates. Although wetlands show structural and functional overlap with terrestrial and aquatic systems, they often serve as the interface between these 2 systems. Wetland structure, internal critical processes, and ecosystem functions are sufficiently different from terrestrial and aquatic systems to require a knowledge base specific to wetlands. We provide here only a brief discussion of certain unique aspects of wetland ecosystems. The reader is encouraged to review relevant published literature for a more complete foundation in wetland ecology. Recommended readings include Ethrington (1983), Mitsch and Gosselink (1993), and NRC (1995).

Wetlands can best be viewed as complex temporal and spatial mosaics of habitats with distinct structural and functional characteristics. Variation in vegetation structure represents one of the most striking examples of spatial and temporal pattern in wetland habitat. Depending upon the type of wetland, the system may be dominated by emergent herbaceous or woody macrophytes, with open water relegated to relatively small areas among blades of emergent plants or to small open patches within the emergent stand. However, regardless of the dominant vegetation, horizontal zonation is a common feature of wetland ecosystems, and in most wetlands, relatively distinct, often concentric bands of vegetation develop in relation to water depth. Bottomland hardwood forests and prairie pothole wetlands provide excellent illustrations of zonation in 2 very divergent wetland types (Figures 4-12 and 4-13).

Wetlands may display dramatic temporal shifts in zonation patterns in response to changing hydrology. Entire systems may even shift, for example, between predominantly emergent and open water zones. In periods of little or no water, some



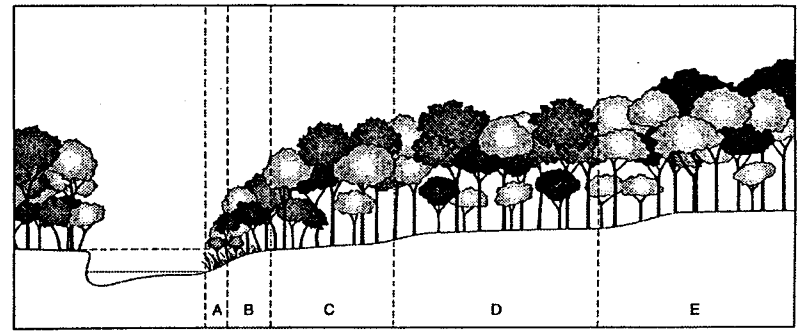


**Figure 4-12** Vegetation zones along South Skunk River, IA (and similar-sized rivers in central hardwoods forest region). A–B) Deposition bank with A) herbaceous plants and tree seedlings grading to B) dominance by *Salix interior* and young *Salix nigra* and *Populus deltoides*. C) Floodplain with maturing *Salix nigra*, *Populus deltoides* and *Acer saccharinum*. D) First terrace dominated by *Celtis occidentalis*, *Junglans nigra*, and *Fraxinus pennsylvanica*. E) Second terrace dominated by *Quercus macrocarpa* and/or *Acer nigrum* depending on soil type and aspect. In larger river bottoms, area C is much expanded with relatively less of areas D and E.

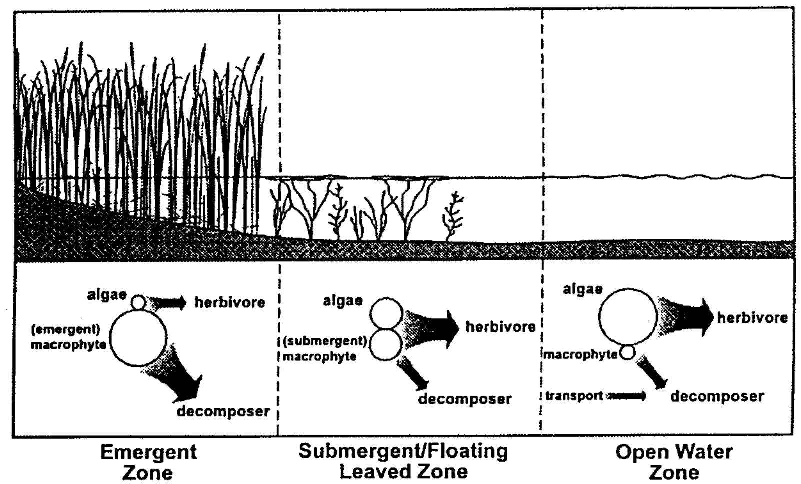


**Figure 4-13** Spatial pattern in vegetation and energy flow in prairie pothole wetlands

wetlands may temporarily become almost terrestrial in form and function. Yet, the same system in other years or in other seasons of the same year may be flooded to the extent that the system becomes, in small or significant part, largely aquatic in nature. Temporal patterns are in fact important characteristics of many wetland types. Seasonal cycles are a major feature of floodplain forests, for example. These systems are flooded during winter and spring periods of high stream flow and bankfull discharge but are typically dry by mid to late summer due to drainage and ET. Longer-term cycles are a major feature of prairie pothole wetlands, which undergo dramatic, more or less cyclic changes in response to a variety of environmental factors including water-level fluctuations and grazing (van der Valk 1989; Mitsch and Gosselink 1993). As a result, these systems may exhibit major year-to-year variations in vegetation structure and distribution and in the relative importance of vegetated and open water zones (Figure 4-14).

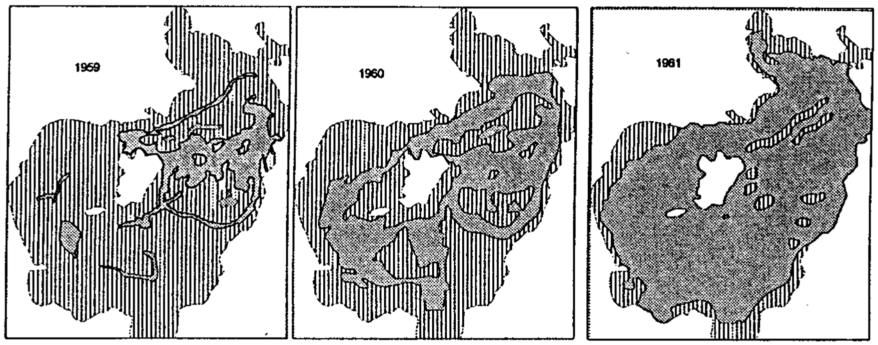


**Figure 4-14** Annual changes in open water (shaded) and emergent vegetation (hatched) in a prairie pothole wetland (reprinted with permission from University of Notre Dame, Weller and Spatcher 1965)

Given the complex temporal and spatial structure of wetlands, it is important to understand the critical habitat characteristics that exert control over major aspects of wetland function. In comparison to our understanding of vegetation dynamics, there is relatively little information regarding the influence of vegetation on wetland environments. However, it is clear that vegetation structure has dramatic effects on the physical, chemical, and biological attributes of wetland habitats (Carpenter and Lodge 1986; Rose and Crumpton 1996). Wetland macrophytes affect environmental attributes and biogeochemical processes in a variety of ways, including reducing light available to algae and/or submersed macrophytes, reducing water temperatures (due to shading), reducing circulation of the water column with effects on gas exchange and material transport, increasing inputs of detrital carbon, enhancing transport of gases to and from the sediment (rhizosphere), and either reducing or enhancing mineral uptake and release. In addition to direct and indirect effects on biogeochemistry (see Chapter 3), vegetation structure is one of the most important factors affecting foodweb structure and bioenergetics in wetland ecosystems. Despite the obvious oversimplification, it is useful to distinguish 3 broad classes of primary producers in wetlands with regard to foodweb dynamics:

1) emergent macrophytes,
2) submergent and floating leaved macrophytes, and
3) planktonic and periphytic algae.



Emergent macrophytes are similar to terrestrial plants in that their biomass is high in structural components such as cellulose and lignin. Their leaves and stems have the low nutrient content and high carbon-to-nitrogen ratios typical of terrestrial plants of similar growth form, and their food value is relatively low. In general, herbivory on emergent macrophytes is very low, and most of their production is transferred to the detrital pool. Nonetheless, the impact of herbivore activity may be extensive at times. For example, the complete destruction of emergent vegetation by muskrats in freshwater marshes has been documented numerous times (van der Valk 1989). However, even during these events, muskrats prefer roots and shoot bases and rarely consume leaves and stems of emergent macrophytes. These tougher materials are instead discarded or used to build lodges, thus entering the detrital pool. Due to the prevalence of structural compounds such as cellulose and lignin, detritus derived from emergent macrophytes is relatively resistant to digestion or decomposition, especially under anaerobic conditions. Nutrient content is even lower and carbon-to-nitrogen ratios higher than in the living plants, and as a result, decomposition frequently requires nutrient subsidy from external sources such as chemical fertilizers.

In contrast to emergent macrophytes, submergent and floating leaved macrophytes have substantially less structural material. Their tissues generally have higher nutrient content and lower carbon-to-nitrogen ratios. Due to their higher nutrient content, the food value of submergent and floating leaved plants can be relatively high in comparison to emergent macrophytes. Herbivory on submergent and floating leaved macrophytes is highly variable, but in comparison to emergent macrophytes, a larger portion of their production may be consumed by herbivores rather than being transferred directly to the detrital pool. The principal herbivores consuming submergent and floating leaved macrophytes include waterfowl, macroinvertebrates, and fish. Due to the relative paucity of structural compounds, detritus derived from submergent and floating leaved macrophytes is relatively labile and relatively easily digested or decomposed.

Planktonic and periphytic algae, of course, have very little structural material. Their tissues have very high nutrient content and low carbon-to-nitrogen ratios. Algae have very high food value and are easily consumed and digested by a wide range of herbivores including microzooplankton, macroinvertebrates, and fish. Although grazing rates vary, much of the algae produced in wetlands is consumed by herbivores rather than being transferred directly to the detrital pool, significantly more than in the case of emergent or submergent macrophytes. Detritus derived from algae is very labile and easily digested or decomposed.

Most freshwater wetlands are assumed to be dominated to a lesser or greater extent by a food chain that is weblike and detritus-based (Mitsch and Gosselink 1993). However, based on the preceding discussion, it is clear that spatial heterogeneity in vegetation structure can result in a mixture of detritus-based and producer–herbivore-based food webs (Figure 4-13). For example, emergent macrophytes dominate production in the emergent zone of freshwater marshes. Most of this production could be expected to enter the detrital pool, with relatively little consumption by herbivores. In contrast, phytoplankton dominate production in the open water zone of freshwater marshes, and much of their production would probably be consumed directly by herbivores. In wetland zones dominated by submergent and floating leaved macrophytes, these macrophytes and their attached algae might both contribute significantly to total production. In either case, a significant proportion of the total production would probably be consumed directly by herbivores. Given these relationships, it is probably better to characterize the food webs of freshwater marshes and most other wetlands not as either detritus-based or producer-herbivore based, but rather as complex mosaics of habitats with distinct food webs. It is important to understand that seasonal as well as longer-term shifts in habitat mosaics and in their associated food webs and biogeochemistry are fundamental aspects of the character of many wetland ecosystems (Figure 4-14).

## Applying the Ecological Factors to a Wetlands-specific Risk Assessment

As part of the data collection for the risk assessment, keep in mind that, as a general rule, ecotoxicological or other types of tests that might be applicable for coastal or marine wetlands may not be suitable for freshwater wetlands and vice versa (Kent et al. 1994). It is incumbent on those using any of the tests or undertaking the laboratory or field studies to fully understand their applicability, limitations, and interpretation.

The ecosystem approach given here was constructed to maximize flexibility in approaching the risk assessment, made necessary by the diversity of freshwater wetlands that may be encountered, in addition to the multitude of factors or stressors that may be at work in the particular wetland under study (Kusler and Kentula 1990; Zentner 1994). Figure 4-15 provides a simple hypothetical illustration of the stressors or factors at work in a wetlands at 2 different times to explain that the magnitude of these stressors is highly dynamic. This figure further emphasizes that all forms of stressors, biological, chemical, and physical, are integrated within the overall risk faced by ecological receptors, such as wetlands, and that the interlinkage of these stressors must be understood and recognized when conducting a risk assessment (Kentula et al. 1993).

An ecosystem approach stresses the key concept of interlinkage of the wetland components (NRC 1992, 1995). An additional overarching provision is that the approach to data collection and evaluation should be tiered (or phased) so that resources are focused effectively and there is ample opportunity for the risk assessor and risk manager to discuss the scientific and policy implications as the risk assessment proceeds (USEPA 1994a, 1997).



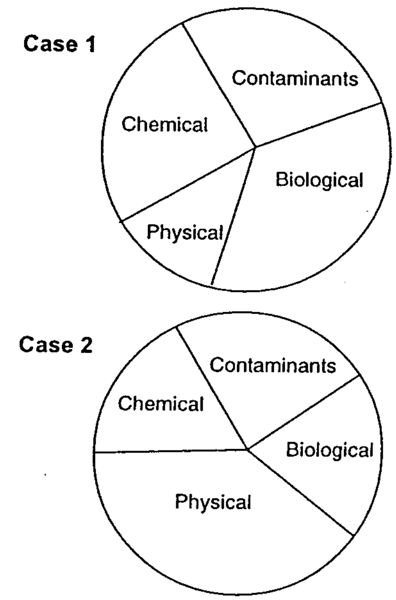


**Figure 4-15** Main groups of stressors in the environment. Note that the relative proportions of the stressors to the whole stress are dynamic, both temporally and spatially, and not evenly divided.

### Problem formulation

There are several main points to consider when beginning a wetland risk assessment. First is to gather and review previously developed information such as aerial photographs, historical maps, land use documents, previous biological or chemical studies, etc. Also important is to gain an understanding of the hydrology and geology driving the wetlands under study. For example, is the wetland riparian, a prairie pothole, or another type? As noted earlier, wetlands vary in their structure primarily due to hydrological and geological conditions, both of which will influence the focus of the risk assessment (NRC 1995) as well as the data collection process. Another key aspect is to determine or define the spatial extent of the area under study. For some wetlands, this will amount to only a few acres; for others, it may encompass an entire watershed of several thousand acres or more.

An early step in problem formulation, with respect to wetlands, is the use of wetland evaluation models (e.g., Brinson 1993; Bartoldus et al. 1994), which can help establish the important characteristics of the wetland under study. More importantly, however, these and other models can be useful when discussing assessment and measurement endpoints.

## Development of assessment and measurement endpoints

One of the most important steps in the problem formulation phase is establishing clear assessment endpoints because they set the stage for all of the forthcoming effort. Assessment endpoints specific to freshwater wetlands can vary tremendously due to the diversity of potential wetland types that may be encountered and due to the myriad functions the wetland may serve. In a diversion from the general practice in ecological risk assessment, under the proposed USEPA framework (USEPA 1992) the assessment endpoints may or may not be biologically or ecologically based. For example, the hydrology, geomorphology, soils, and other aspects of the wetlands may be far more important a focus than some of the biological resources (Brinson 1993). This is not to suggest that ecologically based endpoints are not important, but that they entail abiotic as well as biotic considerations. In fact, directing the risk assessment at the ecosystem or landscape level requires recognition of the abiotic and biotic components and their linkage.

Some important values and functions of freshwater wetlands, from which assessment endpoints can be derived, are shown in Table 4-6 (see also Brinson 1993; Bartoldus et al. 1994; Richardson 1994). These are not exhaustive but can be used as starting points in the risk assessment. Examples of possible assessment endpoints specific to freshwater wetlands are shown in Table 4-7.

**Table 4-6** Important values and functions of freshwater wetlands

| Value or function | Mechanism or activity |
|---|---|
| Hydrological | |
| Flood protection | Water storage and control |
| Water quality | Sediment control; nutrient production or export |
| Ecological | |
| Habitat | Vegetative growth and maintenance |
| Human | |
| Recreation | Fishing, hunting, wildlife watching |
| Commercial | Fishing, timber harvesting |

**Table 4-7** Possible assessment endpoints for freshwater wetlands

| Assessment endpoint | Significance |
|---|---|
| Hydrological | |
| Maintain natural supply of water to wetland | Key to maintaining proper level of hydration |
| Provide sediment control | Reduces turbidity and sediment loading to nearby waterbodies |
| Geomorphological | |
| Maintain bank stability | Reduces erosion of stream and river banks |
| Ecological | |
| Maintain level of primary productivity | Underpins food web stability |



A hypothetical case helps illustrate the development of assessment endpoints and the shift in focus of the risk assessment. Assume that the freshwater wetland under study is one that is dependent on a constant supply of high-quality ground water. In this example, one assessment endpoint might be to protect the supply of high-quality ground water to the wetlands by preventing exposure of the ground water to nonchemical stressors (physical diversion of the ground water for other purposes). Albeit an oversimplification of an actual situation, this illustrates a shift from ecologically driven assessment endpoints to hydrologically driven endpoints. However, as stressed in the ecosystem approach given here, the ecology, hydrology, and geomorphology are inseparable and, in fact, define the ecosystem being protected. In this situation, protection is afforded against that which poses a risk to the sustainability of the wetland: loss of ground water, without which the wetland ceases to exist.

In one of the most extensive ecological risk assessments conducted in a wetland environment, the Clark Fork River (CFR) Superfund Site in Montana (Pascoe and DalSoglio 1994; Linder et al. 1994; see also Chapter 5), none of the assessment endpoints for the riparian wetlands or the river itself included protection of the water supply per se. This does not imply that the risk assessment was done incorrectly, but that the primary focus was to protect ecological resources (primarily plants and animals) versus the one key component responsible for the wetlands themselves: water. Further, this does not mean that there were no important biologically driven assessment endpoints, but that the assessment endpoints for wetlands should include other parameters that are crucial to the long-term sustainability of the wetland itself.

Under the hypothetical case described above, another important consideration could be ensuring that the ground water is meeting a minimum "quality" standard. Quality could be defined as a particular range of pH, turbidity, or specific conductance, or as an absence of chemical stressors at some threshold concentration (e.g., dissolved Se concentrations below 5 g/L. Regardless of the situation, it is important to establish the assessment endpoints clearly in the context of what is vital to sustaining or improving the health of the freshwater wetland, recognizing the inseparability of the ecological, hydrological, and geomorphological components.

Likewise, measurement endpoints may or may not have a biological or ecological basis. Nevertheless, they must be relevant to and linked directly with the assessment endpoints. In the hypothetical case, measurement endpoints may be analytical determinations of contaminant concentrations in the water supplying the wetlands, the specific conductance or suspended solids levels in the water, the flow of water to the wetlands, and others. Implicit, too, is the understanding that biologically driven assessment and measurement endpoints may be included as well.

Numerous endpoints can be used to assess impacts to biological functions. Following is a synopsis of some key biological measurement endpoints for wetland risk assessment (Table 4-8).

**Table 4-8** Important hydrogeomorphic, biogeochemical, ecological, and compound-specific parameters for assessing exposure in freshwater wetlands (field and/or laboratory measurements)

| Hydrogeomorphic information | Biogeochemical information | Ecological information | Compound-specific information |
|---|---|---|---|
| Type of water input (capillary, precipitation, etc.) | Soil-sediment origin and characterization | Plant communities | Volatility |
| Type of water flow (surface, subsurface, etc.) | Microbial activity | Aquatic and benthic community structure | Hydrophobicity |
| Type of water outputs (percolation, evaporation) | Oxidation/reduction conditions | Wildlife survey | Water solubility |
| Suspended-sediment load and characterization | OM content of sediments | | Octanol/water partition coefficient |
| Sedimentation rate | | | Hydrolysis<br>Photolysis<br>Biodegredation |

## Methods and endpoints for wetlands

While numerous field and laboratory methods are available for evaluating aquatic habitats and sediments within wetlands, relatively few are available for testing wetland soils. Sources of information regarding aquatic and sediment contamination evaluation are listed below, and only more recently developed soil test methods will be summarized here for use in wetlands risk assessment.

Whether qualitative and reliant on published information or quantitative and implemented as part of a designed study, aquatic field surveys and biological tests for evaluating wetland risks can be achieved by evaluating biological effects associated with chemical, physical, or biological stressors. Frequently, these tools are used in the measurement or monitoring of wetland populations and community structure through structural endpoints such as relative abundance, species richness, community organization (diversity, evenness, similarity, guild structure, and presence or absence of indicator species), and biomass. Functional endpoints, such as cellular metabolism, individual or population growth rates, and rates of material or nutrient transfer (e.g., primary production, organic decomposition, or nutrient cycling) are less commonly measured. While functional measurements are important in interpreting the significance of an observed change in population or community structure, functional measures are difficult to interpret in the absence of structural information, have not been standardized, and require considerable understanding of the system and processes involved.



### *Species richness and relative abundance*

Species richness (the number of species in a community) and relative abundances (the number of individuals in any given species compared to the total number of individuals in the community) are structural endpoints commonly measured in field surveys of periphyton, plankton, macroinvertebrates, and fish regardless of whether the habitat is a wetland or flowing surfacewater feature. Estimates of relative abundance or species richness can yield readily interpretable information on the degree of contamination of wetland habitat (Pascoe et al. 1994). Loss of a particular species can be critical when that species plays an important role in a community or ecosystem (Karr et al. 1986).

### *Biomass*

Biomass measurements, defined as the mass of tissue present in an individual, population, or community at a given time, are another potential structural endpoint critical to wetland risk assessment. As summarized by LaPoint and Fairchild (1989), biomass can be directly measured gravimetrically on wet or dry tissue. For example, biomass may be estimated gravimetrically by using pooled samples of individuals or by an indirect method, e.g., invertebrate or fish biomass can be indirectly estimated by using empirical or published length:weight regressions. Biomass of periphyton communities is also commonly measured. Measurements of phytoplankton or periphyton biomass can be estimated on the basis of ash-free dry mass (AFDM) or chlorophyll *a* content (APHA 1992). Chlorophyll measurements are performed by solvent extraction, followed by spectrophotometry or fluorometry (APHA 1992).

### *Indicator species*

The presence or absence of indicator species is commonly used to assess adverse effects to ecological communities (Karr et al. 1986; Hilsenhoff 1988; Plafkin et al. 1988). While originally derived from the saprobian system in which certain species and groups were found to generally characterize stream and river reaches subject to organic wastewaters (Kolkwitz and Marsson 1902; Gaufin 1958; Sheehan 1984), the application of indicator species to wetlands is clearly practiced, e.g., within the delineation process. History has shown that the indicator species concept lacks broad applicability to all types of contaminant stress, however. Furthermore, species selection may occur in aquatic habitats that are chronically polluted with low levels of contaminants over sufficiently long periods. In some wetlands, as well as flowing surface water, the IBI may be pertinent to the risk assessment process.

### *Indices*

Biological indices in wetland risk assessments, as in other ecological risk assessment applications, can be used to mathematically reduce taxonomic information to a single number, or index, to simplify data for interpretation or presentation. Indices can be classified among several types:

1) evenness (measuring how equitably individuals in a community are distributed among the taxa present),
2) diversity (calculating the abundance of individuals in 1 taxon relative to the total abundance of individuals in all other taxa),
3) similarity (comparing likeness of community composition between 2 sites), and
4) biotic indices (examining the environmental tolerances or requirements of individual species or groups).

Although indices may aid in data reduction, they should never be divorced from the actual data on species richness and abundance. Relying on a single index such as the Shannon-Weiner may be misleading for any system at risk, including wetlands. For example, a few individuals evenly distributed among several species could give a relatively high index of diversity, even though a habitat is grossly polluted. In addition, statistical assumptions of independence, normality, and homogeneity of variance are frequently invalid for these derived, proportional measures. Hence, when indices are used, statistical transformations (e.g., arc sine) or rank-order statistics are recommended (Siegel 1956; Green 1979; Hoaglin et al. 1985).

### *Guild structure*

For wetland communities, data generated at the species level can be analyzed according to guild structure. Guilds, or functional feeding groups, are classifications based on the manner in which organisms obtain their food and energy. Invertebrates can be classified among such functional groups as collector-gatherers, piercers, predators, scrapers, and shredders (Merritt and Cummins 1984; Cummins and Wilzbach 1985); and fish can be classified as omnivores, insectivores, and piscivores (Fausch et al. 1984; Karr et al. 1986). Avian communities in wetlands are increasingly being analyzed within the context of guild structure (Adamus 1993a, 1993b). Shifts in community guild structure may reflect changes in the trophic-dynamic status of a wetland. For example, contaminant impacts on a wetland may eliminate or reduce periphyton and thus concomitantly reduce the relative abundance of scrapers (herbivores) in relation to other invertebrate guilds such as collector-gatherers. Effects must be fairly strong to assess changes in guild structure. For contaminant studies in wetlands, community and guild analysis should also be supported by physical habitat and chemical information, since these may alter production and dynamics of biological populations and, consequently, confound the interpretation of wetland community data. Needless to say, the selection of appropriate reference locations is critical to wetland assessments that incorporate community and guild analysis.



### *Plankton*

Many devices are available for sampling plankton for their enumeration and analysis. Sampling techniques for phytoplankton and zooplankton are similar in various surfacewater habitats. The choice of an individual sampling technique, sample size, and sample numbers, whether for zooplankton or phytoplankton, will depend upon the characteristics of the aquatic habitat (in terms of depth, density of organisms, and spatial variation).

### *Macroinvertebrates*

Benthic invertebrates are the most common fauna used in ecological assessments of contaminants, whether sediments are in wetlands or other surfacewater habitats. Numerous excellent references deal with the collection, identification, and analysis of benthic invertebrate populations (e.g., Southwood 1978; APHA 1992). Typical measurement endpoints include relative abundance and species (or taxon) richness. Trophic guild structure can be determined from taxonomic identifications to species (Merritt and Cummins 1984; Cummins and Wilzbach 1985). Indices of diversity, evenness, and community similarity can also be calculated. In any given contaminant effects study, careful consideration must be given to the comparability of samples among stations.

### *Fish*

In biological monitoring and evaluation, as well as in wetland risk assessment, fish may be recommended for use because

1) regulators and the public can easily understand the implications of the effects of pollution on fish;
2) fisheries have economic, recreational, and aesthetic values;
3) the identification of fishes is relatively easy (compared to that of micro- and macroinvertebrates);
4) the environmental requirements of fish are well known; and
5) fish are perceived as "integrators" of effects at lower trophic levels (Hendricks et al. 1980).

However, the size, distribution, and response of freshwater fishes are sometimes difficult to quantify because variations in spatial distribution and year classes are large (Lagler 1978). Additional difficulties in the quantification of fish populations are caused by the selectivity and efficiency of the sampling gears used (Hendricks et al. 1980). However, consideration of these factors can allow unbiased comparisons of different wetland habitats that support their being considered as part of the wetland risk assessment process.

The types of analyses performed on data from the collected fish include relative abundance, species richness, and size structure. One method for fish community assessment is the IBI (Karr 1981; Karr et al. 1986), where IBI is weighted on the basis of individual species tolerances for water quality and habitat conditions. The IBI was developed to determine the effects of decreased habitat quality on fish communities of midwestern streams, but for some wetlands it may be quite applicable to the risk-assessment process. The index is composed of 12 individual metrics divided into the fields of species composition and richness, trophic composition, abundance, and condition. Scores of each metric are classified as "best," "average," or "worst" (each class having a numerical weighting) in relation to reference data (Fausch et al. 1984).

## Sediment and soil methods and endpoints for wetlands risk assessment

While not as readily available as aquatic or sediment toxicity test methods (e.g., Peltier and Weber 1985; Weber et al. 1988), methods have been identified for testing soil biota (e.g., USEPA 1989). For wetlands, the application of biological tests should provide a comparative toxicity database upon which wetland-specific soil evaluations can be completed. Screening (unamended wetland soils yielding percent effect) and definitive tests (amended soils potentially yielding median effective concentrations) may be completed with standardized test species to evaluate toxicity within a biological assessment. Additionally, to assure adequate information for ecological evaluations of soil contamination, species having site-specific relevance may also be tested (Parkhurst et al. 1989). When performed in parallel with standard test methods, these site-specific tests (e.g., using resident plant species) may be diagnostic and indicate biological responses (e.g., development of metal resistance) that are associated with soil exposures. Presently, the application of laboratory bioassays to wetland risk assessment is increasing, particularly in developing biological databases that contribute to the ecological risk assessment process. To enhance the ecological relevance of site-specific biological tests and to reduce the potential extrapolation error associated with interspecific comparisons, use of standard and site-specific test species in ecological assessment should be considered in soil testing (see Linder et al. 1993).

### *Plant test methods*

Plants associated with wetlands have been used extensively to assess water and sediment quality. The wide variety of tests developed has targeted the effects of both water column and sediment-borne toxic materials. The types of aquatic vegetation used for these purposes range from microscopic unicellular algae to relatively large flowering plants. The 3 most commonly applied test methods include chlorophyll *a* concentration, growth, and contaminant uptake.

Growth measurements (biomass accumulation per unit of time) have been widely applied as an assessment method for a variety of freshwater estuarine and marine species. Much of the testing has been conducted on sediments in the laboratory, using unicellular phytoplankton such as *Selenastrum capricornutum* (freshwater) and *Skeletonema costatum* (marine) (e.g., Thomas et al. 1990; Ankley et al. 1993). Until recently, use of rooted wetland macrophyte growth has been limited. Growth is perhaps the least specific measurement endpoint. A response such as reduced



growth rate is not tied to specific sites within the plant where reactions or processes are altered by specific chemicals. This is especially true for rooted macrophytes. The advantage of measuring growth is that it is an integrator of all effects of toxicants on plants, it is relatively easy to measure, there is a wide range of past use, and it can be done with acceptable precision in both the field and laboratory.

The physiology of chlorophyll production and maintenance is quite well known. Chlorophyll occurs in virtually all plants and is the primary pigment involved in the important ecological process of photosynthesis. The correlation between chlorophyll concentration and photosynthetic rate commonly is strong. Chlorophyll concentration relative to contamination of water or soils has been measured in unicellular algae, macrophytes, and periphyton communities (e.g., Bassi et al. 1990). Chlorophyll concentration generally reflects the mass of plant material present, as well as being an indication of the health of the material. Toxicants can affect the chlorophyll molecule directly or through the process of energy transfer during photosynthesis. A method recently applied for determining the effects of toxicants on chlorophyll (and photosynthesis) involves the measurement of delayed fluorescence. The technique appears to be highly sensitive and relatively easy to conduct.

Contaminant uptake by plants has been applied primarily to rooted macrophytes. It is assumed that most of the uptake occurs through the roots and that the concentration of the contaminant compounds in leaf tissues is directly related to the concentration in the soil or sediment. Uptake has received wide application in fresh and marine systems and has been carried out under both laboratory and field conditions (e.g., Kovacs 1978; Lee et al. 1981). Uptake of contaminants relies on several assumptions that must be taken into account for interpretation of results. Chemicals may be modified to form nontoxic compounds by the plant. Certain chemicals are not concentrated, while others are, which may bias the interpretation of what chemicals are present in the test medium. However, these uptake measurements are more relevant for evaluating risks to herbivores (and bioavailability of chemicals in sediment) than for deciding what is there per se. Finally, uptake rates may be inhibited by the toxicity of other materials in the medium, and the test organism may be inhibited in its ability to accumulate the contaminants.

While measurements of plant growth, chlorophyll content, and contaminant uptake are the most commonly used methods, several other are in various stages of development and implementation. These methods include measurements of photosynthetic rate, chloroplast morphology, peroxidase activity, root growth, seed germination, seedling growth, and reproduction.

The strongest approach to the assessment of wetland subsystems may be to use a combination of several methods to evaluate contamination of water and sediments. This combination would indicate both ecological and physiological responses of the plants to the media and would increase the power of the analysis through verification of responses using several endpoints.

### Seed germination and root elongation

Techniques modified from methods originally developed in the plant and weed science disciplines have yielded short-term tests that assess toxic chemical effects on plants. The seed germination and root elongation bioassays are laboratory toxicity tests that directly and indirectly assess toxicity of soils and evaluate toxicity endpoints (seed germination and root elongation) pertinent to ecological assessments for terrestrial and wetland habitats. Seed germination tests measure toxicity associated with soils directly, while root elongation tests consider the indirect effects of water-soluble constituents which may be present in site samples. These methods have been used extensively in soil contamination evaluation, including a comprehensive wetlands risk assessment (Linder et al. 1994; Pascoe and DalSoglio 1994; Pascoe et al. 1994).

### Rooted aquatic plants

Wetland soils frequently complicate standard methods for phytotoxicity assessment, owing to the saturated character of their soils. Wetland soils may resemble sediments in many respects, particularly when seasonal or ephemeral climatic conditions alter soil water-holding capacity, which may confound interpretations of germination and growth responses in standard plant testing species (e.g., buttercrunch lettuce, *Lactuca sativa*). Standardized rooted aquatic plant toxicity tests, however, have been developed and should be considered on a site-specific basis for hydric soils and freshwater or estuarine sediment evaluations. The most well-developed method uses *Hydrilla verticillata*, but additional test methods using sago pondweed (*Potamogeton pectinatus*) may also be valuable in evaluating wetland soils or sediments (Byl and Klaine 1991; Fleming et al. 1992).

### Laboratory evaluations with wetland and upland plants

Freshwater marsh plants may be used to evaluate sediments or hydric wetland soils as outlined by Walsh et al. (1991). The method was originally designed to test single toxicants or defined chemical mixtures in defined media, but it can be modified to test field-collected sediments or wetland soils that may be appropriate to wetland risk assessment. In general, the method utilizes rooted marsh plants and evaluates the effects of contaminated soils and sediments on early seedling growth and survival. For example, *Echinochloa crusgalli* is one species of marsh plant specifically identified in the test procedure, but alternative marsh plants (e.g., *Spartina alterniflora*) may be identified on a site-specific basis and tested, provided the selected plants are amenable to the test format outlined.

Primarily in response to the assessment needs associated with land disposal of dredging materials, the U.S. Army Corps of Engineers Waterways Experiment Station (WES) has developed a test method for evaluating phytotoxicity and bioaccumulation potential in a freshwater plant, the yellow nutsedge (*Cyperus esculentus*). The method is applicable to wetland risk assessments and can be used in either flooded wetland or upland habitats. From an ecological perspective, the test evaluates toxicity endpoints (e.g., growth) that may directly relate to field observations regarding plant cover or vegetative vigor (WES 1989; Folsom and Price 1992). It is also useful for evaluating bioaccumulation of contaminants in the diet of herbivores.



### Alternative test species in seed germination, root elongation, and early seedling survival and vegetative vigor tests

In these tests, measurement endpoints are frequently similar (e.g., growth, germination), but the species being tested differ. In part, these differences reflect soil matrix characteristics that might limit the success of any given test system, especially in wetland soils. For example, lettuce seed is frequently used in seed germination tests, but some soils may not be amenable to testing with a domesticated species selected for optimal growth in a particular soil matrix. Contaminant effects and matrix effects may potentially be confounded when the life history characteristics of a test species preclude or potentially limit its usefulness in any given phytotoxicity test method. Additionally, for interpretation of wetland-specific ecological effects, the support of a comparative toxicity database may be insufficient within a risk assessment context. Thus, more relevant test species may be beneficial to evaluate ecological effects with a wetlands risk assessment, and measurement endpoints (e.g., survival and growth) used to evaluate relationships between ecological indicators and soil toxicity may be considered using methods modified for tests with alternative species. For example, methods to evaluate seed germination using various species of plant seeds (agricultural crops, vegetables and herbs, flowers, and trees and shrubs) are briefly summarized by the Association of Official Seed Analysts (AOSA) in their Rules for Testing Seeds (1990). Here, exposure conditions specific to various species are tabulated, including suggested substrates and optimum incubation temperatures for germination testing as well as test duration specifications. Furthermore, special pretreatment of native seeds, e.g., prechilling or scarification, is also specified, and methods for distinguishing between nongerminated seeds and nonviable seeds are identified (e.g., tetrazolium and embryo excision tests). On a wetland-specific basis, these alternative test species may be more conducive to ecological interpretation, especially when soil matrix effects unique to wetlands can potentially confound contaminant effects on seed germination and emergence.

### Soil biota biomass and diversity

Without question, wetlands are complex biological systems, and wetland soils are critical components in the characterization process. A thorough consideration of the methods applicable to wetland soils characterization with a risk assessment setting is beyond our present scope. However, wetlands functions and processes are clearly dependent upon a healthy soil. For example, nutrient cycling would not occur without organisms to perform the majority of the critical processes. Soil organisms perform many wetland processes, and in unimpacted soil, there usually (but not always) are several organism groups that perform any particular process. For example, the dependency of vegetation on the presence of mycorrhizal fungi and on a functional soil–organism nutrient cycling system may be quantified within a wetlands risk assessment, and evidence is accumulating that at least some plants are dependent on symbiotic organisms for establishment or survival (Reeves 1985; Janos 1987). Clearly, other measurement endpoints could be identified (Linder et al. 1992), and while not exhaustive, methods are available to evaluate these within the context of wetland risk assessment:

1) bacterial biomass and community structure,
2) fungal biomass and community structure,
3) protozoan diversity, and
4) nematode diversity and community structure.

***Solid-phase and aqueous-phase Microtox***

While aqueous-phase testing with Microtox has been readily available for 10 to 15 years, solid-phase testing has only recently been commercially available (Microbics 1992). As previously summarized (Warren-Hicks et al. 1989), Microtox relies upon measurements of bioluminescence for an evaluation of a sample's toxicity. The test, whether aqueous- or solid-phase, utilizes freeze-dried cultures of the marine bacterium *Photobacterium phosphoreum* and is based on the inhibition of bioluminescence by toxicants (Bulich 1979, 1982, 1986). The results of several studies of pure compounds and complex chemical mixtures suggest that aqueous-phase testing with Microtox generally agrees with standard fish and invertebrate toxicity tests (Curtis et al. 1982). Solid-phase testing with Microtox, however, does not have a comparable database established for developing statements regarding its correspondence with standard soil tests using, for example, earthworms.

***Earthworms tests***

While not applicable to all wetland soils, earthworms have become a primary test organism for soil contamination evaluations. From an ecological perspective, earthworms are significant in improving soil aeration, drainage, and fertility (Edwards and Lofty 1972), although the comparative database does not unequivocally suggest that earthworm toxicity measurements are reflective of soil health. To enhance the ecological relevance of site-specific biological tests and to reduce the potential extrapolation error associated with interspecies comparisons, testing with site-specific species should be considered in soil evaluations. The earthworm bioassay most frequently used is a modification of a method described by Goats and Edwards (1982) and Edwards (1984) and uses lumbricoid earthworms as the test species. *Eisenia foetida* may be used in these tests because it is easily cultured in the laboratory and reaches maturity in 7 to 8 weeks at 25 °C. *E. foetida* is responsive to a wide range of toxicants, and the comparative database suggests that similar toxicity responses can be anticipated regardless of the subspecies being tested (Neuhauser et al. 1986).



***Nematodes tests***

Soil-inhabiting nematodes represent one of the most readily available soil invertebrates that should be studied during soil contamination evaluations within an ecological effects assessment for a wetland. Owing to their usually high numbers, their role in soil decomposition processes, and their significant contribution to soil nutrient dynamics (e.g., dispersion and grazing on microflora, potential stimulation of bacterial activity, and promotion of nutrient mineralization), soil nematodes directly as well as indirectly reflect the health of the wetland soil. *Panagrellus redivivus* has a relatively well-developed literature in aquatic toxicity testing (Samoiloff et al. 1980) and has been used for evaluating single chemicals and complex chemical mixtures (Samoiloff et al. 1983), including applications to sediment evaluations. Most frequently, *P. redivivus* has been used in conjunction with other biological assessments (e.g., *Daphnia magna* or *Ceriodaphnia dubia* testing) for evaluations of water quality, but the test system has also been applied to sediment toxicity testing (Samoiloff et al. 1983). Work with *P. redivivus* has been well described in the comparative toxicity literature, but another, more recently developed nematode test using *Caenorhabditis elegans* (e.g., van Kessel et al. 1989; Williams and Dusenbery 1990) may be applicable for ecological effects assessments. *P. redivivus* and *C. elegans* tests measure acute—lethal—and subacute or sublethal effects related to growth, reproduction, and mutagenicity. Both methods are short-term tests and generally require less than 4 to 5 d for completion, although long-term tests that measure reproductive effects (e.g., number of offspring) may require 7-d exposures.

Unlike *P. redivivus*, *C. elegans* is a native soil-dwelling nematode (Briggs 1946 as cited by van Kessel et al. 1989), and tests with this nematode may more closely reflect soil contaminant effects in terrestrial habitats. Williams and Dusenbery (1990) studied the toxic effects of metals in aqueous solutions using *Caenorhabditis elegans*, and in their comparative analysis, *C. elegans* acute toxicities (LC50s) for single-compound metal exposures complemented and were consistent with acute toxicity results from *Daphnia magna* and sediment macroinvertebrates. As suggested by various authors (e.g., Popham and Webster 1979; Haight et al. 1982; Doelman et al. 1984; van Kessel et al. 1989), for some toxicants like heavy metals, the existing toxicity database for nematodes was developed, and extending these methods to soils should be considered within ecological effects assessments. For example, while the testing with either *P. redivius* or *C. elegans* was originally developed for testing surface water or sediment pore waters, nematode tests are directly applicable to evaluating soil extracts or interstitial waters.

***Arthropods (insects) tests***

Various methods have been developed for evaluating chemical effects on terrestrial insects, especially pesticide effects on nontarget species (e.g., USEPA 1982), and these methods are directly applicable to wetlands risk assessment. As ecological indicators of soil contamination, terrestrial insects, and soil arthropods in general, are potentially critical targets within an ecological effects assessment. Within ecological contexts, terrestrial invertebrates play a role in communities and ecosystems that involves integrated functions such as decomposition, grazing, predation, and pollination (Croft 1990). While methods that evaluate adverse biological effects in terrestrial invertebrates exposed to soil contaminants are not widely considered in the ecological effects assessment process at present, their contributions have increased and should continue to increase in the near future, especially for wetlands risk assessment. Through strategies similar to those used with aquatic invertebrates (e.g., Plafkin et al. 1989; Klemm et al. 1990), terrestrial insects would be amenable to soil contaminant evaluations for wetlands, particularly given field survey information regarding insect community structure and population numbers in wetlands at risk. For example, to evaluate soil microarthropods quantitatively and qualitatively, techniques are readily available to extract, enumerate, and identify these organisms in reference and impacted soil samples. Soil microarthropods are easily extracted from the soil using Tulgren high-efficiency extractors (e.g., Seastedt and Crossley 1980; Anderson 1988 ). The extracted organisms can then be counted using dissecting microscopes and identified to genus, or form-group. Recent innovations in computer-assisted identification (Hypercard) have also reduced the time required to identify these organisms (Moldenke et al. 1991).

#### *Terrestrial arthropod (non-insect) and isopod tests*

Outside of North America, terrestrial arthropods other than insects have been considered from the perspective of accidental or coincidental exposure to potentially harmful chemicals (Croft 1990). While not exclusively focused on wetlands, these methods are directly applicable to the risk assessment process for wetlands. For example, to evaluate effects of agrichemical pesticides or biological control agents on nontarget invertebrates, laboratory methods have been standardized for evaluating chemical effects on mites (e.g., Sewell and Lighthart 1988). While terrestrial arthropod tests methods are few and present a limited history in ecological effects assessments for wetlands, their role in the environment (Croft 1990) requires that these organisms should receive consideration as ecological receptors during the risk assessment process. The methods developed for pesticide evaluations could be directly applied to wetland soils contamination evaluation. Alternatively, soil-derived eluates could be used in the testing process, if the study design indicated that indirect routes of exposure were likely to occur, e.g., nonpoint source runoff into wetlands from agricultural lands. While a variety of test species have been used in the standard tests developed in Europe and the United States (Hassan 1985; Hassan et al. 1987; Croft 1990), the laboratory test methods using non-insect arthropods are relatively straightforward and easily could be modified to directly meet the requirements of a soil contaminant evaluation for wetlands.

Similarly, biological assessments using terrestrial isopods have historically been considered in soil contamination evaluations, although standardization, e.g., through the American Society for Testing and Materials (ASTM) or the Organization for Economic Cooperation and Development (OECD), is lacking. As field indicators of contaminant exposure, the isopod literature suggests that whole-body and organ-specific contaminant bioaccumulation may be monitored with these animals, particularly for some environmental chemicals, e.g., metals (Beyer et al. 1984; Beyer and Anderson 1985; Hopkin 1986, 1990).



#### *Mollusk tests*

Wetlands are habitats that are frequently impacted by hazardous waste disposal sites, and mollusks are often regarded as representative invertebrates characteristic of these habitats (Pennak 1978). Coincident with these habitat-related questions, some families of freshwater mussels (Unionidae) have been identified as critical species for ecological risk assessments for some environmental chemicals (agrichemicals) (USDOI 1989). Accordingly, methods potentially amenable to ecological effects assessments at Superfund sites have been developed to evaluate chemical effects and acute toxicity for sensitive life stages in various mollusk species (Johnson 1990). In contrast to concerns regarding habitat loss and contaminant effects on freshwater mollusks, efforts to develop effective molluscides have also yielded test methods (e.g., Getzin and Cole 1964; Crowell 1979) that may be applicable to the ecological assessment needs for wetlands. Historically, marine and estuarine mollusks have been used in toxicity and ecological effects assessments within the Office of Pesticides Program (USEPA 1995), and these methods could be equally applicable to contaminant-related questions for wetlands risk assessments. Analogous tests with freshwater mollusks have recently been developed. For example, the Unionidae mollusks are characteristic freshwater mussels, and numerous species could be considered within a toxicity assessment setting. In developing a freshwater mussel test, *Anodonta imbecilis* was initially selected as a representative unionid mollusk; however, the techniques described by Johnson (1990) should be applicable for testing mussels with similar reproductive strategies. Most frequently, the tests involve the early developmental stages of the mussel, or glochidia, and juvenile mussels, depending upon endpoints being measured. Guidance for developing the test with freshwater mussels followed ASTM E729 (1997b), and while not widely used at this time, toxicity assessments with freshwater mussels should be considered within an ecological effects assessment for wetlands.

In contrast to the freshwater mussel test that was primarily developed in response to ecological risk assessment questions related to agrichemical use, test methods that evaluate terrestrial snails and slugs were developed as efficacy tests for evaluating molluscides (e.g., Getzin and Cole 1964; Crowell 1979). These methods, however, are readily adapted for wetland risk assessment.

#### *Amphibian test methods*

Wetlands are habitats that are frequently impacted by hazardous waste sites, and evaluating and monitoring these transition zones between upland and surfacewater areas will require a variety of field and laboratory techniques (Tiner 1984; Adamus and Brandt 1990). Amphibians—frogs and salamanders—may be representative of the fauna potentially critical to ecological effects assessments for wetlands. Amphibian test systems are standardized through ASTM (T29 1997b, E1439 1997c). Early embryos of the African clawed-frog (*Xenopus laevis*) are used in the standardized test; however, much work has been completed with alternative test species and should be considered on a site-specific basis (e.g., Linder et al. 1990; ASTM E1439 1997c; Linder, Wyant et al. 1991).

## Interplay of risk management and risk assessment

Important to all risk assessments, whether for wetlands or terrestrial environments, are the early discussions held between the risk assessor and the risk manager. These should define the scope, timing, level of effort, and constraints involved with the risk assessment. There will need to be resolution of issues specific to freshwater wetlands, and the particular type of wetland, between the risk manager and risk assessor before any work is begun.

This discussion may have several important outcomes. First is agreement on the spatial extent or magnitude of the wetland. Small, easily managed wetlands may require a reduced or screening-level assessment to satisfy the requirements of the risk manager. On the other hand, wetlands that are tens or hundreds of acres, that reside in the midst of major industrial activities, or that are complex in terms of their hydrology, soils, geomorphology, etc. may require a much greater level of effort on the part of the risk assessor. In this latter situation, landscape and ecosystem issues arise and can readily complicate the effort. For example, some wetlands may be dependent on source water outside of the study area, or for that matter, in another state, region, or watershed. Like a number of stressed wetlands in North America, the wetland may be vitally important in controlling floods in a particular area but may not represent a highly valuable habitat (e.g., a *Phragmites* sp.-dominated wetlands) (Bartoldus et al. 1994).

It is also important for the risk manager and the risk assessor to decide on the important stressors and receptors that will be the focus of the assessment. As data are collected and evaluated, additional stressors and receptors may become evident and may justify a realignment of the focus. A confounding issue that often arises at this time is whether the risk assessment will take a multi-stressor or single-stressor approach. It is rare that only a single stressor will be present, yet to approach the risk assessment using multiple stressors requires advancement beyond current science. Today there is inadequate understanding of how to deal with multiple stressors only qualitatively because there is no recognized, validated method for integrating impacts from multiple stressors. Thus, without a clear understanding of what is driving the risk management decision and of the regulatory and jurisdictional issues, the risk assessor may be left with insufficient or at least unclear guidance.



## Exposure assessment

Inputs of chemical and nonchemical stresses to freshwater wetlands occur through geological, biological, and hydrological pathways typical of other ecosystems (Mitsch and Gosselink 1993). Geological input from weathering of parent rock, although poorly understood, may be an important source of exposure in some wetlands. Biological inputs include photosynthetic uptake of C, N fixation, and biotic transport of materials by animals. Except for gaseous exchanges such as C and N fixation or aerial deposition, however, inputs to wetlands are generally dominated by hydrology. Hydrologic transport to freshwater wetlands may occur through precipitation, surfacewater flow, or groundwater flow. The hydrologic exposure pathways of freshwater wetlands are determined by their flooding regime or by the balance between precipitation and evapotranspiration.

Hydrodynamics will affect exposure levels in both the aquatic and soil-sediment compartment of a wetland, as it will to a large extent determine the soil-sediment chemistry by producing anaerobic conditions, importing and removing OM, and replenishing nutrients. Exposure can occur in transition zones between the wetland and surrounding upland areas. It is important to consider this area as well when examining potential exposure scenarios.

Ideally, exposure in the wetland ecosystem is assessed based on representative monitoring data. In the absence of measured data, exposure can be predicted in the context of a wetland-specific hydrogeomorphic, biogeochemical, and ecological setting. In the case of a chemical exposure assessment, information on the inherent properties of substances should be used in combination with the wetland characteristics in order to derive exposure concentrations or levels. Describing the level and distribution of a stressor in the wetland environment and its changes with time (e.g., in concentration or chemical form) is a complex process and needs to include a rigorous evaluation of what drives exposure. In order to ensure that predicted aquatic and sediment exposures are realistic, all available knowledge of the wetland ecosystem should be integrated in the exposure evaluation of a chemical stressor. Some measurements or parameters that can be important when evaluating or predicting exposure of chemical and/or nonchemical stressors in freshwater wetlands are listed in Table 4-8.

Compound-specific information and biogeochemical processes affecting exposure in the different compartments are usually derived and extrapolated from standard laboratory tests or literature data. Applicability of literature data and data from standard tests to freshwater wetland ecosystems requires review and, ideally, field verification.

## Biological assessment

Defined earlier, biological assessments are primarily ecotoxicological tests performed in either a field or laboratory setting. While there are many issues related to the conduct and application of ecotoxicological tests (Levin et al. 1989), they represent one of the main sources of effects information available to the risk assessor. It is beyond the scope of this section to detail the methods or protocols for these tests. However, the publications cited in Table 4-4 include standard testing protocols as well as those developed through the auspices of the OECD.

Once the key stressors and receptors have been identified, the biological assessment should consider toxicity to wetland organisms or plants in the overlying water as well in the sediments, provided the stressor is likely to enter and persist in the sediments. In addition, the assessment may need to extend to the transition zones surrounding the wetlands because some stressors will impact adjacent terrestrial environments. These areas should be evaluated only if there are clear, potential pathways for exposure of receptors. Because the primary focus of the biological assessment should be at higher levels of organization, the risk assessor should be cognizant of which tests or series of tests are designed to measure population-, community-, or ecosystem-level effects. Furthermore, the endpoints of the test, whether lethality, reproductive impairment, growth, etc., should be understood and their linkage to the assessment endpoints clearly defined before any work is begun.

Depending on their scope, biological assessments in the aquatic environment could include representative, and ideally sensitive, species of

1) primary producers,
2) primary consumers,
3) microbial community,
4) saprophages or detrivores, and
5) carnivores.

Potential tests for the primary producers could include tests with algae and vascular plants, both submerged and emergent forms. Effects on primary consumers could be evaluated by testing representative species of protozoa, invertebrates, insects, and amphibia. Inhibition of microbial activity, important in wetland's nutrient recycling and transport, could be evaluated by studying the effect on aerobic and/or anaerobic respiration. Toxicity tests with crustacea and insects can be used to assess effects on the saphrophages/detrivores community. Finally, standard acute and chronic tests are available to assess effects on fish.

Biological assessments of the benthic communities should take into account pathways of exposure. In addition, observed effects will be strongly influenced by sediment-soil biogeochemical conditions such as organic carbon content, particle size distribution, sulfide content, redox potential (RP), and time period allowed for equilibration to occur between dissolved and sorbed fractions of chemical stressors (USEPA 1990). Available test methods concern detrivores or mixed detrivores/herbivores/carnivores and include insect, annelida, and crustacea species with both acute and chronic endpoints (USEPA 1990).



Recently, the OECD reviewed aquatic testing methods for pesticides and industrial chemicals (OECD 1995). The review included both pelagic and benthic test methods. An overview of the recommended test methods—applied to freshwater wetlands—is shown in a foodweb frame in Figure 4-16.

It was recommended by OECD that the guidelines and tests take the form of a framework for taxonomic groups rather than for single species, whenever possible. This should make it possible to test representatives from different wetland compartments and facilitate extrapolation of obtained test results to the wetland of interest. Furthermore, the guidelines and tests should include both acute and subchronic or chronic toxicity endpoints, depending on the assessment endpoints.

Most of the impacts on freshwater wetlands will occur in the aquatic environment, i.e., the sediment and overlying water. Even so, the terrestrial environment surrounding or transitioning to the freshwater wetland may also be at risk, depending on the type of stressor and the exposure. Species that are dependent on the wetland structure and function (e.g., insects, amphibians, reptiles, small mammals, and birds, and transition-zone plants, trees, and shrubs) should be considered when potential effects are evaluated. Standardized toxicity tests are currently available for many insects, some amphibians, and numerous small mammals and birds, but few have been adapted for the species most often associated with freshwater wetlands. Acute and chronic bioassays with rodents and lagomorphs have been used for many years to determine the toxicity of chemicals and other materials that may also pose a risk to humans. Similarly, standard acute and chronic tests with species of waterfowl and upland birds have been widely used in the field of environmental toxicology.

There are, however, few tests that have been developed for nonfood plants, although the tests currently used in regulatory programs for pesticides and herbicides may be useful. For example, tests for root elongation and shoot development, seed germination, and other methods are known and may be useful in evaluating toxicity of soils in the transition zone. Other soil tests, some using earthworms, might be useful in this context. Keep in mind that the primary focus of the assessment is the wetland itself, and it is there that the effort should begin.

Unfortunately, few tests lend themselves easily to determining the potential toxic effects on trees and shrubs that may inhabit the transition zones. In those situations, it may be more plausible to determine impacts in situ on those trees and shrubs located adjacent to the wetlands of concern. Methods developed by forestry scientists (e.g., measuring growth rate, stand composition, and overall vigor) can be utilized for this.

Using standardized toxicity tests brings up several important considerations, some of which are mentioned in Chapter 9. One of these concerns data interpretation and is driven primarily by the fact that most easily maintained species used in testing are not the same species generally found in freshwater wetlands. Thus, the uncertainty of extrapolating from one species to another within the same genus could be as large

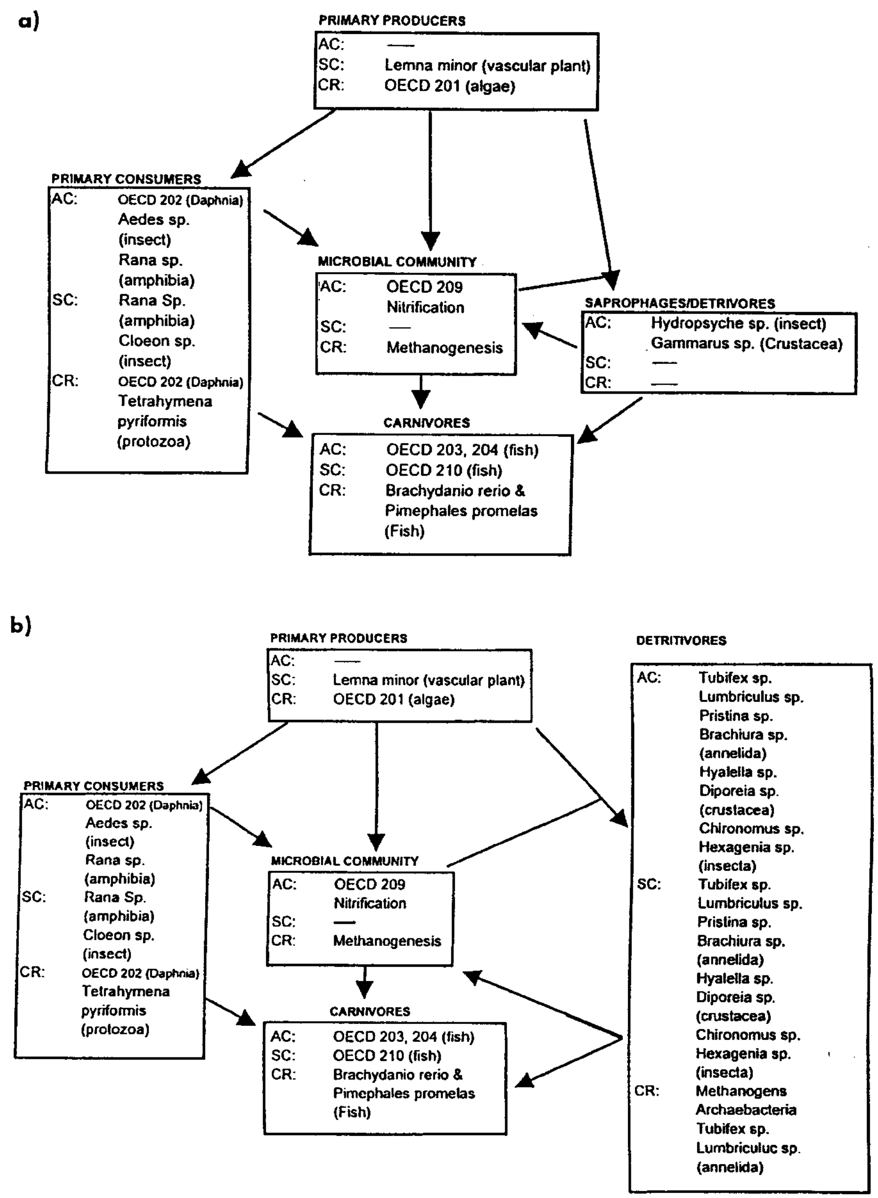


**Figure 4-16** Taxonomic grouping of test organisms recommended for freshwater wetland risk assessment by the OECD (1995): a) Primary producer-herbivore-carnivore food web, b) Detritus-based food web. AC= acute tests, SC= subchronic tests, CR= chronic tests



as extrapolating from rodents to humans. Therefore, it is important to understand the limitations of surrogate species testing and its application to risk assessment. Other uncertainties arise when acute exposure test data are extrapolated to chronic exposure situations, high concentration-response studies to low-concentration exposures, laboratory to field results, and others. All of the results from the biological assessment should be taken in context with other data that will be developed as part of the risk assessment.

Selection of biological tests for wetland ecotoxicity evaluation should be driven by the exposure assessments affected by the hydrogeomorphic and biogeochemical characteristics of the wetland of interest.

## Ecological assessment

Ecological assessment primarily determines the impacts of stressors at the population, community, or ecosystem level. In general, standardized ecotoxicology tests do not lend themselves to this type of assessment, and few provide useful ecosystem-level information (Kelly and Harwell 1989; Cairns and Niederlehner 1992). In addition, there are significant temporal and spatial issues that come into play. Measuring a significant change in an ecosystem or at the landscape level may require years or decades of study, yet the risk assessor and risk manager are faced with a much more compressed time line. Just as important, it is difficult to isolate easily studied areas of the wetlands from the surrounding ecosystem that supports it, which may require the risk assessor to include caveats and large uncertainties in the risk assessment.

Given this situation, most ecological assessments are field studies that measure structural components of the ecosystem, including the size and make-up of the habitat, the biomass or standing crop of important plants and animals, and the abundance and diversity of plants and animals. There are, however, functional measurements (Bartoldus et al. 1994; Richardson 1994) that might be useful in understanding the ecological integrity of the wetland. For example, wetlands are extremely important to biogeochemical processing and nutrient cycling (e.g., N and P) (NRC 1995; Chapter 3, this volume) as well as in primary productivity and C, N, P export (Chapter 2, this volume). These functional aspects of wetlands, often considered to be indicative of ecosystem-level processes, depend heavily on microbial communities, water flow, benthic macroorganisms, and other parameters (Brinson 1993). As a result, these functions may be important areas for the risk assessor to consider when designing and conducting the ecological assessment, especially when the assessment focuses on effects at the ecosystem level. Similarly, population- or community-based measures may be useful, provided they have a direct relationship to the assessment endpoints and have been validated scientifically.

Net primary productivity and carbon or energy flow also offer wetland processes that may be measured to assess ecosystem-level effects, provided the measures are integrated across the entire wetland. In this situation, the measure is made of a wetland's net product, resulting from an integrated, interconnected process.

Often, results of the biological and ecological assessments can become inputs to various trophic-level or foodweb models. Such models can give the risk assessor a useful tool to develop a refined conceptual model of how stressors could impact the various processes in the wetlands. The problem with some of the trophic-level or foodweb models is that they require a substantial amount of data, preferably site-specific in nature, lest the uncertainty remain high. Given that fact, the risk assessor and risk manager should decide early on whether the size of the wetland or the complexity of the problem warrants such data-intensive assessments.

## Evaluation of Case Studies using the Ecosystem Framework

### In retrospect: Would ecosystem-based wetland planning have altered the outcome of the Kesterson episode?

Kesterson Reservoir (see Chapter 6) provides a case history that can be used to assess how well the ecosystem approach performs in evaluating risks associated with proposed wetlands. Limited availability of water was the key issue driving the development of Kesterson's wetlands. Since the 1890s, diversion of water for agricultural use had taken a tremendous toll on the quantity of wetlands remaining in the San Joaquin Valley of California. By the 1970s, when Kesterson was developed, the view generally held by wetland managers in the valley was that any water was better than no water. Viewed in hindsight, the rationale for this thinking is clearly flawed because of water quality issues such as selenium contamination, but at the time, there was no equivalent wetland from which to draw information. However, had an environmental planner been present using the ecosystem approach, would the resultant risk assessment have effectively identified and predicted the problems that eventually occurred?

In order to answer this question, we must look at the basic components of the ecological framework (Figure 4-2). A key factor indicated in the assessment process for Kesterson would have been to thoroughly characterize the water sources and hydrologic regime, i.e., quantity and quality of irrigation drainage, in the context of the arid climate present at the site. Had this step been performed adequately, several key pieces of information should have emerged to guide the decision process. First, it should have been apparent that the evaporative nature of the climate would maximize the likelihood that salts and chemical contaminants in the water source could become concentrated in the wetlands. Second, knowing that the intended water source was subsurface irrigation drainage and not fresh water, adequate chemical characterization would have been indicated. A water quality analysis would have revealed the presence of elevated concentrations of Se, B, and, in some instances, As or other elements. Even though much of the toxicity database that now exists for these trace elements would not have been available then, it should still have been clear that the water source contained an atypical concentration of salts and trace elements. This, in turn, would have signaled a risk factor that required further investigation. The ecological framework would have indicated to the planner that thorough biological effects testing was necessary to determine whether the water source was acceptable for developing the wetland to meet its primary goal, i.e., as habitat for migratory waterfowl and shorebirds. Carrying out these effects studies would have quickly revealed the toxic hazards from trace elements and indicated that irrigation drain water should not be used to develop Kesterson.



The critical failure in the Kesterson episode was lack of recognition that water quality is a primary consideration in wetland development. Kesterson also illustrates the difficulty of using 1 wetland to achieve 2 objectives. In the case of Kesterson, these were wildlife habitat and disposal of irrigation drainage. Clearly, these were not compatible objectives from the standpoint of water quality. The ecological framework to risk assessment could have identified this problem early in the planning stage and recommended steps to avoid the wildlife toxicity problems that eventually developed.

### Current evaluation: Application of the ecosystem framework to risk assessment at Milltown Reservoir Wetlands

The work at Milltown Reservoir Wetlands (MRW) (see Chapter 5) illustrates the strengths and limitations of an integrated ecosystem-based approach to ecological risk assessment. This work at MRW also illustrates how the approach, when applied within a risk assessment context, provides resource managers with tools that would enhance their decision-making process and minimize or at least clearly identify sources of uncertainty. At MRW, the ecosystem approach outlined in this chapter clearly provided a framework for minimizing the heavy-metal–related problems that have developed and are being evaluated throughout the MRW-CFR watershed today. For example, at MRW, land-use and water-use planning was poorly implemented in the up-front siting of the construction project for the hydroelectric facility located at the confluence of the Blackfoot and Clark Fork Rivers of western Montana. This historic, and in many instances current, practice of pursuing widespread land-use and water-use practices with only limited forethought for the interconnectedness within ecological systems is a serious flaw that quickly becomes apparent when the ecosystem-based approach is applied. Whether these resource-use practices are mining, agriculture, forestry, or recreation oriented, various environmental problems have arisen throughout the western U.S. in the absence of an ecosystem-based approach to risk assessment.

Using MRW as our example, the initial decision to site a hydroelectric facility at the Hellsgate of the Clark Fork just east of Missoula, MT might have been reconsidered, especially if the watershed had been more fully characterized and appreciated. For example, the relationships between the upstream source areas near Anaconda and

Butte were clearly not understood at the turn of the century when the hydroelectric facility was constructed at Milltown. If an analysis of the hydrology (surface and subsurface) as well as the geomorphology had been completed as part of the current problem formulation phase of the risk assessment process, the facility might have been constructed at an alternative location, or other measures to reduce sedimentation behind the dam would have been considered.

The current problems from metals and arsenic associated with the soils and sediments are a direct consequence of an incomplete analysis of the surface and subsurface hydrology within the CFR watershed. While this criticism is retrospective, the history of the MRW nonetheless reinforces the value that the ecological risk assessment framework offers to resource managers today. Again, using MRW as it looks today, the available risk analysis for the wetland clearly indicates that the present and near-term risks are low relative to metal- and As-related questions in the wetland, and the focus of attention upstream from the reservoir is well deserved from a management perspective. Here again the ecosystem-based approach has served decision-makers well, and while more subtle issues remain regarding incompletely answered questions (e.g., regarding rhizosphere exposures in the wetland), within a risk assessment context, sufficient information was available to address the current and near-term issues related to the wetland. More importantly, the uncertainty associated with these decisions was more clearly understood and characterized in the ecological risk assessment for the wetlands at Milltown Reservoir, primarily because of the risk analysis activities indicated by the framework. Even in the comprehensive ecological risk assessment for MRW that is currently available, incomplete knowledge is apparent. However, when pursued within an ecosystem context, the uncertainties associated with those data gaps were manageable within the near-term and long-term plans for the wetland and the CFR watershed.

As the work at MRW illustrates, environmental contaminant problems in wetlands often are not a simple problem of chemicals alone, but instead are a complex set of interconnected issues that involve a large noncontaminant component. More often than not, habitat alteration has provided an equal, if not greater, contribution to a multiple stressor setting for resources at risk like those at MRW. Within the ecosystem-based approach, the ability to distinguish between and among various stressors will be required more frequently in resource management decisions that are focused on low-concentration exposures to environmental contaminants and the potential subacute effects that may result. While our present state-of-the-science achieves varying degrees of completeness for any particular risk assessment, the ecosystem approach clearly supports a decision-making process that will minimize uncertainty and potentially yield resource management decisions that are dynamic and achievable in the near and distant future.



## In the future: Will the Everglades restoration be successful? A challenge of hydrological, chemical, and biological linkages

Restoration of the Everglades involves several policy, partnership, and technical challenges. The policy and partnership issues are beyond the scope of this chapter, but the technical issues that will influence policy decisions are clearly on-point here. Successful Everglades restoration will hinge on the ability of scientists to integrate the concepts discussed in this chapter and provide consensus advice to decision-makers. If this integrative approach is not used, the end result will be a lack of environmentally sound management policies, i.e., an even bigger disruption of the natural wetland ecology than now exists.

From the time the earliest explorers came to south Florida, the challenge was how to drain the region so that productive use could be made of the land. These efforts began in earnest during the 1880s with the work of Hamilton Disston and his projects to connect Lake Okeechobee to the Caloosahatchee River. By the mid-20s, the state had completed the main north–south canals from the lake to the coast, and agriculture became an important part of the region's economy. By 1970, the USACOE completed the major components of its Central and Southern Florida Project, which linked all the drainage canals and water management structures into a comprehensive water management system. The project has been completely successful in meeting its major objectives of flood control, agricultural water supply, and protection of urban well fields from saltwater intrusion.

However, the project, and the 5 million people now able to live in the region, is producing unexpected side effects. The wading-bird population of south Florida has diminished to less than 10% its level of 50 years ago. Florida Bay is experiencing vast algal blooms, which are killing sponge and seagrass beds vital to shrimp and fish populations living there. Nutrient runoff from dairy, citrus, and sugar farms around Lake Okeechobee is transforming the river of sawgrass to large cattail communities. Citizens and government are looking for ways to restore much of the lost biological function in the Everglades.

For the past 20 years, scientists and technical managers have examined the problem from within their areas of expertise. The water managers worked on the water management problems. The land-use managers worked the land-use problems. The chemists and toxicologists studied the effects of the various chemicals and nutrients, using their established protocols and approaches. The biologists studied various biological problems, but usually in a very narrow context rather than with an ecosystem perspective. The only clear agreement from all of these investigations is that the altered (drained) system is causing a series of effects that no one understands very well.

The progress that is needed will depend on linking the analysis in ways suggested in the section entitled "The ecosystem approach: integrating ecology, hydrology, geomorphology, and soils of wetlands." For example, the analysis of nutrient effects on sawgrass cannot be complete without an appropriate analysis of different hydrological alternatives. The source of elevated mercury levels in the food chain is a vexing dilemma. Altered hydrology is cited as one of many possible causes, with a restored hydrology proposed as the solution. However, the changes in water quality parameters other than Hg, as well as shifts in plant and animal life that would likely accompany these hydrological modifications, must also be considered. The soils in the region vary greatly, and engineers routinely design water-control structures based on their compatibility with the soil conditions. However, less studied and understood are the possible influences of changing water regimes on biology and groundwater hydrology. Most importantly, land-use assumptions and decisions will continue to have a decisive impact on all of these analyses and outcomes.

The critical challenges in south Florida will be to develop an ecosystem approach and a landscape view to our science. Both of these areas represent critical gaps in our knowledge, but both are the focus of current initiatives to adjust our approach. Without an ecosystem approach, the information is incomplete and consensus is impossible. Without a landscape view, the issues become intractable and solutions impossible. The ecological framework to risk assessment allows scientists to examine the issues in a context that can provide the consensus necessary for success.

## Research Needs and Recommendations

Previous ways of assessing wetlands have been expanded into the ecosystem approach outlined in this chapter. This approach integrates ecology, hydrology, geomorphology, and soils of wetlands for the evaluation of impacts and risks from chemical, biological, and physical stressors. When the ecosystem method to wetlands-specific risk assessment was applied, it became apparent that there is a need to establish and implement a consistent operational framework in order to make full use of this approach. Several concerns are evident. The effect of multiple stressors (chemical, physical, and biological, of anthropogenic or natural origin) must be an integral component of the assessment process. Standardization of reliable acute, subchronic, and chronic tests is necessary. Alternative exposure–effects scenarios must be evaluated. Understanding fate and transport of chemicals and their interaction with physical, chemical, and biological toxicity-modifying factors is critical. The parameters that must be measured on-site to determine potential pathways and fate of toxins need to be better quantified. There are also specific information needs for organismic, population and community, and ecosystem levels of organization.

### Organismic

The levels of uncertainty resulting from presently used, standardized toxicity tests have not been carefully scrutinized in the context of freshwater wetland ecosystems. For example, plant toxicity data are generally based on one green alga *(Selenastrum capricornutum, Scenedesmus* sp. or *Chlorella* sp.) and one vascular aquatic plant



(duckweed species, *Lemna minor* or *Lemna gibba*), but there may be a need to represent different groups of photosynthetic and nonphotosynthetic wetland organisms. Using a species battery approach could lessen the potential errors associated with interspecies extrapolation. This is also true for micro- and macroinvertebrates. Laboratory-to-field extrapolations of single-species tests may therefore be improved by using ecologically relevant species batteries with subsequent field validation.

### Population and community

As with interspecies comparisons, errors and uncertainty associated with interorganizational extrapolation need to be evaluated. This would include scaling issues associated with transitions between different levels of biological organization.

### Ecosystem

The ecosystem approach proposed here uses HGM characterization together with wetland functions as the criteria for establishing transport, fate, and effects of both chemical and nonchemical stressors. Coupled with toxicity assessments at 3 organizational levels—organismic, population and community, and ecosystem—this approach may be used to describe exposure and effects of stressors in freshwater wetlands, both as a predictive tool and to describe existing conditions. Practical application of the approach will provide a better understanding of how physical, chemical, and biological factors modify the intensity of the stressors. Tools for integrating and analyzing these complex ecosystem interactions need to be refined or, in some cases, still need to be developed. Approaches for evaluating the influence of seasonal and spatial variability are especially necessary.

Toxicity assessments involve tests of varying complexity (single-species, mesocosm, ecosystem assessments, etc.). As a rule of thumb, costs escalate with increasing complexity and single-species laboratory bioassays being the least expensive. From a cost-benefit perspective, the least complex test that can adequately predict ecosystem effects should be the method of choice, providing proper validation has been carried out. The ecosystem approach may reduce the overall cost of risk assessment by identifying key biological, chemical, and physical parameters that must be evaluated early in the assessment process.

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